The thematic assessment report on

INVASIVE ALIEN SPECIES AND THEIR CONTROL

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THE IPBES THEMATIC ASSESSMENT REPORT ON INVASIVE ALIEN SPECIES AND THEIR CONTROL

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The thematic assessment report on

INVASIVE ALIEN SPECIES AND THEIR CONTROL

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IPBES is an independent intergovernmental body comprising over 140 member Governments. Established by Governments in 2012, IPBES provides policymakers with objective scientific assessments about the state of knowledge regarding nature and the contributions it provides to people, as well as options for actions to protect and sustainably use these vital natural assets.

The Assessment of Invasive Alien Species and their Control was initiated by a decision from the IPBES Plenary (decision IPBES-6/1) at its sixth session (IPBES 6, Medellin, Colombia, 2018), based on the scoping report (annex III to decision IPBES-4/1) approved by the Plenary at its fourth session (IPBES 4, Kuala Lumpur, Malaysia, 2016). The Assessment was produced in accordance with the procedures for the preparation of the Platform's deliverables set out in annex I to decision IPBES-3/3.

The Assessment of Invasive Alien Species and their Control was considered by the IPBES Plenary at its tenth session (IPBES 10, Bonn, Germany, 2023), which approved its summary for policymakers, and accepted its chapters. All material can be found here: https://www.ipbes.net/ias

FOREWORD

A key objective of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is to provide Governments, the private sector and civil society with scientifically credible and independent up-to-date assessments of available knowledge for better evidence-informed policy decisions and action at the local, national, regional and global levels.

The thematic Assessment of Invasive Alien Species and their Control, or "Invasive Alien Species Assessment" in short, is part of a series of reports whose production was initiated during the "first work programme of IPBES, 2014-2018" and concluded during the current "IPBES rolling work programme up to 2030". The Invasive Alien Species Assessment has been carried out by a multidisciplinary team of 86 selected experts from all regions of the world, including early career fellows, assisted by about 200 contributing authors. More than 13,000 scientific publications were analyzed as well as a substantive body of Indigenous and local knowledge. Its chapters were accepted, and its summary for policymakers was approved, by the IPBES Plenary composed of 143 member States at its tenth session held from 28th August to 2nd September 2023 in Bonn, Germany.

The Invasive Alien Species Assessment builds on the landmark IPBES Global Assessment Report on Biodiversity and Ecosystem Services launched in 2019. The Global Assessment identified invasive alien species as a one of the five main direct drivers of biodiversity loss, with 1 million species of plants and animals now at risk of extinction.

The Invasive Alien Species Assessment explores how invasive alien species affect nature and people globally. It analyzes the status and trends of alien and invasive alien species in all regions of Earth, and identifies major pathways for and drivers of the introduction and spread of such species between and within countries. The Assessment also assesses the

effectiveness of management actions across scales and in various contexts. The Invasive Alien Species Assessment finally outlines key responses and policy options for the prevention, early detection, and effective control of invasive alien species and mitigation of their impacts in order to safeguard nature, nature's contributions to people and good quality of life.

The Invasive Alien Species Assessment highlights that invasive alien species are a major and growing threat to nature, nature's contributions to people, with, in some cases, irreversible changes to biodiversity and ecosystems. Invasive alien species also dramatically impact the economy, food security, water security and human health, sometimes adding to marginalization and inequity. The Assessment demonstrates that with sufficient resources, political will, and long-term commitment, preventing and controlling invasive alien species are attainable goals that will yield significant long-term benefits for people and nature.

As the Chair and the Executive Secretary of IPBES, we wish to recognize the leadership and dedication of the co-chairs, Prof. Helen Roy (United Kingdom), Prof. Aníbal Pauchard (Chile), and Prof. Peter Stoett (Canada) and the hard work and commitment of all the coordinating lead authors, lead authors, review editors, fellows, contributing authors and external reviewers, and to warmly thank them for contributing their time and ideas freely to this important report. We would also like to recognize the hard work and dedication of Naoki Amako and Noriko Moriwake, heads of the technical support unit for this Assessment, Tanara Renard Truong, assessment coordinator, and Ryoko Kawakami, administrative officer. Our thanks go also to the current and former members of the Multidisciplinary Expert Panel (MEP) and of the Bureau who provided guidance as part of the management committee for this report, and to members of the IPBES secretariat including those of other technical support units within the IPBES secretariat, who have supported the production of this report, and its successful launch in the media. We would also like to thank all Governments and other institutions that provided financial and in-kind support for the preparation of this Assessment. We are profoundly aware that work was made more challenging over the past couple of years because of the COVID-19 pandemic which prevented the experts from meeting and connecting in-person as planned, and which created very difficult personal circumstances for many. We express again our deepest thanks and recognition to all involved, on behalf of IPBES.

The Invasive Alien Species Assessment provides the best-available evidence, critical analysis and options for governments, civil society, Indigenous Peoples and local communities, the private sector and all those seeking to address the issue of biological invasions. The Assessment is also expected to support sharing of information within and across countries and capacity building globally. It is our sincere hope that this Assessment will support the implementation of the Sustainable Development Goals of the 2030 Agenda for Sustainable Development (especially Goal 15) and form a significant contribution to the implementation of the Kunming-Montreal Global Biodiversity Framework of the Convention on Biological Diversity, and especially of its Target 6.

Ana María Hernández Salgar
Chair of IPBES (2019-2023)

Anne Larigauderie
Executive Secretary of IPBES

STATEMENTS FROM KEY PARTNERS

Humanity has been moving species around the world for centuries. This practice has brought some positives. However, when imported species run rampant and unbalance local ecosystems, indigenous biodiversity suffers. As a result, invasive species have become one of the five horsemen of the biodiversity apocalypse that is riding down harder and faster upon the world.

While the other four horsemen – changing land- and sea-use, over exploitation, climate change and pollution – are relatively well understood, knowledge gaps remain around invasive species. The IPBES Invasive Alien Species Report is a welcome effort to close these gaps. By providing critical information on trends in invasive species and policy tools to address them, this report can provide a springboard to concrete action on invasive species.

I ask all decision-makers to use this report's recommendations as a basis to act on this growing threat to biodiversity and human well-being – and make a real contribution to achieving the Kunming-Montreal Global Biodiversity Framework by 2030.

Inger Andersen
Executive Director
United Nations Environment Programme
(UNEP)

It is urgent to accelerate efforts against invasive alien species, one of the five major drivers of biodiversity loss that also threatens our health, social development, and culture. UNESCO, as an institutional partner of IPBES, takes pride to have supported this new Assessment Report. It provides a valuable analysis of how invasive alien species are distributed globally and the diverse strategies used to manage them. The report draws on a wide range of knowledge and perspectives from around the world, including Indigenous and local knowledge, which is a central focus of UNESCO's programmes. This crucial information will strengthen ongoing initiatives in UNESCO-designated sites and help decision-makers shape their policies worldwide.

Audrey Azoulay
Director-General,
United Nations Educational,
Scientific and Cultural Organization
(UNESCO)

Invasive alien species pose a substantial threat to livelihoods and food security around the world. They can, for example, manifest as destructive crop or forest pests or displace species targeted by fisheries. They are an important driver of biodiversity loss and hence a threat to the various ecosystem services that support agricultural production and sustainable livelihoods.

The information contained in this report will contribute greatly to efforts to combat the spread of invasive alien species and to meeting Target 6 of the Kunming-Montreal Global Biodiversity Framework. It will be especially valuable for all of us who work to integrate the conservation and sustainable use of biodiversity into the world's agrifood systems to enhance their productivity and resilience.

QU Dongyu
Director-General,
Food and Agriculture Organization of
the United Nations
(FAO)

Invasive alien species – plants, animals or microorganisms that are introduced intentionally or unintentionally into areas where they are not native – remain one of the most striking symptoms of the adverse effect of human activities on our natural world. They not only contribute to wildlife species extinctions, but also pose a rapidly growing risk to progress on the Global Goals – affecting entire ecosystems, economies and food security to human health, wellbeing, and livelihoods.

As anthropogenic factors such as climate change provide the perfect petri dish for alien species to multiply and spread, our decisions and actions must be rooted in a comprehensive understanding of this threat and its future implications.

Addressing this need, this timely analysis by IPBES combines the latest science, data, and new thinking to guide countries, communities, and the United Nations family to prevent, mitigate, and manage invasive alien species, a pivotal step towards advancing the Kunming-Montreal Global Biodiversity Framework targets. That includes leveraging invaluable local knowledge and outlining a range of practical solutions.

This new understanding will allow our global community to take new measures to protect both people and planet from the unwanted and severe consequences of invasive alien species.

Achim Steiner
Administrator,
United Nations Development Programme
(UNDP)

Invasive alien species are one of the five main direct drivers of biodiversity loss globally and the threats they pose to species, to ecosystems and to human well-being are rapidly increasing.

The Kunming-Montreal Global Biodiversity Framework, in its Target 6, aims to tackle the impacts of invasive alien species on biodiversity and ecosystem services, and to reduce the rate of introduction and establishment of invasive alien species by at least 50% by 2030. This is an ambitious target, especially when we consider the increasing levels of global trade and travel.

The IPBES Assessment will provide the best available scientific knowledge to help countries and stakeholders understand and address this growing threat. It will identify tools and policy measures for identifying and regulating pathways of introduction and for eliminating or controlling invasive species that have already been established. Critically, the assessment will take into account different value systems and help to focus actions on priority species, pathways and sites.

Congratulations to IPBES for this critical work. I look forward to seeing its active use by Parties and stakeholders. I believe it will be a critical resource to facilitate the urgent actions necessary to achieve Target 6 and work towards living in harmony with nature.

David Cooper
Acting Executive Secretary
Convention on Biological Diversity
(CBD)

We are indebted to the hundreds of experts, policymakers, and practitioners, including members of Indigenous Peoples and local communities, who generously contributed their time and knowledge as authors, fellows, review editors (all of them listed below) and contributing authors of the Assessment of Invasive Alien Species and their Control, as well as to the management committee who provided oversight and guidance to its development. The Assessment team has contributed thousands of hours of collaborative and voluntary work to provide the best available knowledge on invasive alien species and their control. We have all encountered various challenges, not least the COVID-19 pandemic, throughout the assessment, but the dedication, determination and commitment of everyone involved has been outstanding.

Throughout our time working on the Assessment we have benefited enormously from the invaluable advice, dedication, and constructive contributions from the IPBES secretariat, particularly from the Executive Secretary, Anne Larigauderie, and from Simone Schiele, Bonnie Myers and Hien Ngo, the IPBES Chair, Ana María Hernández Salgar, representatives of member States, and the Multidisciplinary Expert Panel (MEP) and Bureau, especially members of the management committee, Eric Fokam, Shizuka Hashimoto, Rizwan Irshad, Ruslan Novitsky, Rashad Allahverdiyev, Vinod Bihari Mathur, and Youngbae Suh. We have been honored to work with such talented people. The Invasive Alien Species Assessment would not have been possible without the phenomenal contributions and excellent guidance of our technical support unit, headed by Naoki Amako and Noriko Moriwake and supported by Ryoko Kawakami and Tanara Renard Truong during the four years of its production. These colleagues went far beyond expectations, ensuring sustained quality while being thoughtful of and responsive to the needs of the assessment process and our many authors. Further, Tanara is listed as an author on both Chapter 1 and the summary for policymakers, recognizing her incredible contributions to the knowledge and information gathered. We are extremely appreciative of her insights and leadership. We also thank Tom August, Kate Randall and Maro Haas for their skillful and experienced work on data visualization and graphic design. We hugely appreciated the many contributions from Peter Bates who also facilitated collaboration with Indigenous Peoples and local communities. We thank the IPBES communications team for their outstanding work providing expert guidance, training and support through every stage of the Assessment, to ensure the widest outreach of the main

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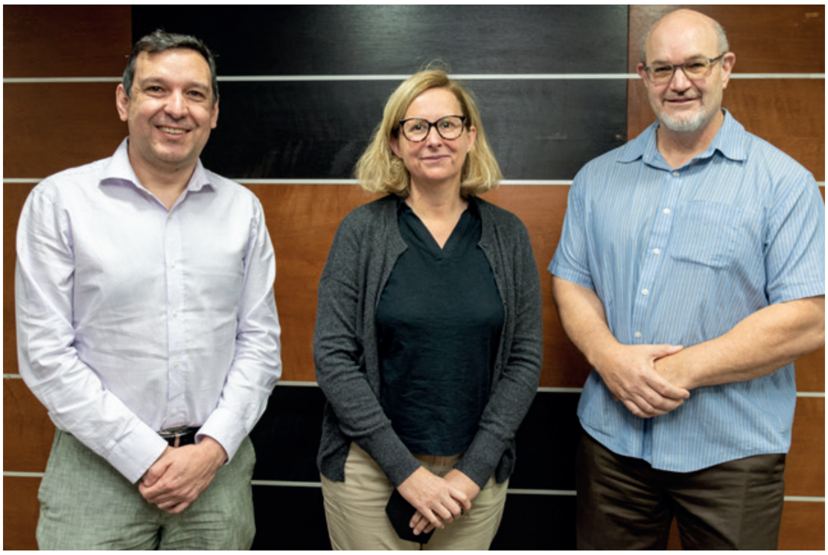

findings of this Assessment. Indeed, the Assessment received media coverage in over 100 countries, with over 5,000 press articles, in the weeks following approval – testament to the meticulous and tireless work of the IPBES communications team.

We are also grateful to the Government of Japan, which generously hosted the technical support unit at the Institute for Global Environmental Strategies (IGES) and the first author meeting, and all the governments who nominated and supported experts. We thank Aarhus University (Denmark) and the University of Concepción (Chile) for hosting our author and/or summary for policymakers meetings. We thank the secretariat of the Convention on Biological Diversity who hosted our first dialogue on Indigenous and local knowledge in Montreal, Canada. We would especially like to acknowledge the support of our home institutions and governments: the UK Centre for Ecology & Hydrology (United Kingdom); the Faculty of Forestry Sciences, University of Concepción and Institute of Ecology and Biodiversity (Chile); and Ontario Tech University (Canada). We have appreciated the encouragement given to us from them and the value they have placed on the work we have been collaboratively leading.

Our gratitude goes to Bureau members Douglas Beard and Sebsebe Demissew Woodmatas, for co-chairing the sessions on the Assessment at the IPBES 10 Plenary (28 August – 2 September 2023, Bonn, Germany) which approved its summary for policymakers and accepted its chapters. We also thank MEP co-chairs Luthando Dziba and Shizuka Hashimoto for their roles in facilitating discussions.

We would lastly like to thank all our friends and relatives who supported us throughout this demanding assessment process. This work would not have been possible without their love, endless support and understanding.

The dedication and contributions of all those mentioned above ensured the outcome of the Assessment of Invasive Alien Species and their Control as a unique, robust and rich report. It has been a privilege to have the opportunity to collaborate with so many inspiring people, in all their many and varied roles. We are confident that the Assessment will be impactful, increasing global awareness of the significant threats to biodiversity and human communities posed by invasive alien species and, very importantly, what can be done to prevent and control them.

Aníbal Pauchard, Helen E. Roy, Peter Stoett
Co-Chairs

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SUMMARY FOR POLICYMAKERS

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1. Authors are listed with, in parentheses, their country or countries of citizenship, separated by a comma when they have more than one; and, following a slash, their country of affiliation, if different from that or those of their citizenship, or their organization if they belong to an international organization. The countries and organizations having nominated the experts are listed on the IPBES website.

DEFINITIONS, CONCEPTS AND THE CONTEXT OF THE ASSESSMENT

The thematic assessment of invasive alien species and their control produced by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) critically evaluates evidence on biological invasions[2] and the impacts of invasive alien species. In alignment with the United Nations Sustainable Development Goals and the Kunming-Montreal Global Biodiversity Framework adopted by the Conference of the Parties of the Convention on Biological Diversity, the assessment outlines key responses and policy options for prevention, early detection and effective control of invasive alien species and mitigation of their impacts in order to safeguard nature, nature's contributions to people, and good quality of life.

For the purposes of this assessment, the terms "native species", "alien species",[3] "established alien species", "invasive alien species", "impacts", "introduction pathways" and "drivers" are represented and defined in **Figure SPM.1**.

The term "biological invasion" is used to describe the process involving the intentional or unintentional transport or movement of a species outside its natural range by human activities and its introduction to new regions, where it may become established and spread.

Species introduced to new regions through human activities are termed alien species. Invasive alien species represent a subset of alien species, being animals, plants and other organisms known to have established and spread with negative impacts on biodiversity, local ecosystems and species. Many invasive alien species also have impacts on nature's contributions to people (embodying different concepts such as ecosystem goods and services and nature's gifts) and good quality of life.[4] Some of the most problematic invasive alien species arrive through multiple introduction pathways and repeated introduction.

Invasive alien species are recognized as one of the five major direct drivers of change in nature globally, alongside land- and sea-use change, direct exploitation of organisms, climate change, and pollution.[5] This assessment considers how biological invasions are facilitated by all those direct anthropogenic drivers, noting that interactions among invasive alien species can enable further biological invasions. The assessment also considers how biological invasions can be influenced by indirect drivers, as identified in the IPBES *Global Assessment Report on Biodiversity and Ecosystem Services:* these include demographic, economic, sociocultural and technological drivers, as well as those relating to institutions and governance. Finally, the assessment considers how biological invasions, and ultimately the impacts of invasive alien species, can be facilitated by natural drivers of change, in particular natural hazards (such as floods, storms and wildfires) and by biodiversity loss itself.

In the context of this assessment, management of biological invasions includes the development of decision support tools; prevention (supported by regulation) and preparedness planning and actions; eradication, containment and control of invasive alien species; site- and ecosystem-based management; and ecosystem restoration.

Other important concepts associated with biological invasion are defined in the glossary of the assessment report. The conceptual basis underpinning the assessment, including the IPBES conceptual framework,[6] and the methodology for reviewing literature are outlined in chapter 1 of the assessment report.

2. This assessment acknowledges that national and local legislation to address biological invasions differ between countries and may include different definitions appropriate to specific national and local contexts.
3. Multiple alternative terms exist to refer to alien species.
4. Annex III to decision IPBES-4/1.
5. IPBES (2019): *The Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Brondizio, E. S., Settele, J., Díaz, S. and Ngo, H. T. (eds.). IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.3831673
6. The conceptual framework for the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services was approved by the Plenary in decision IPBES-2/4 (2013) and updated in decision IPBES-5/1 (2017).

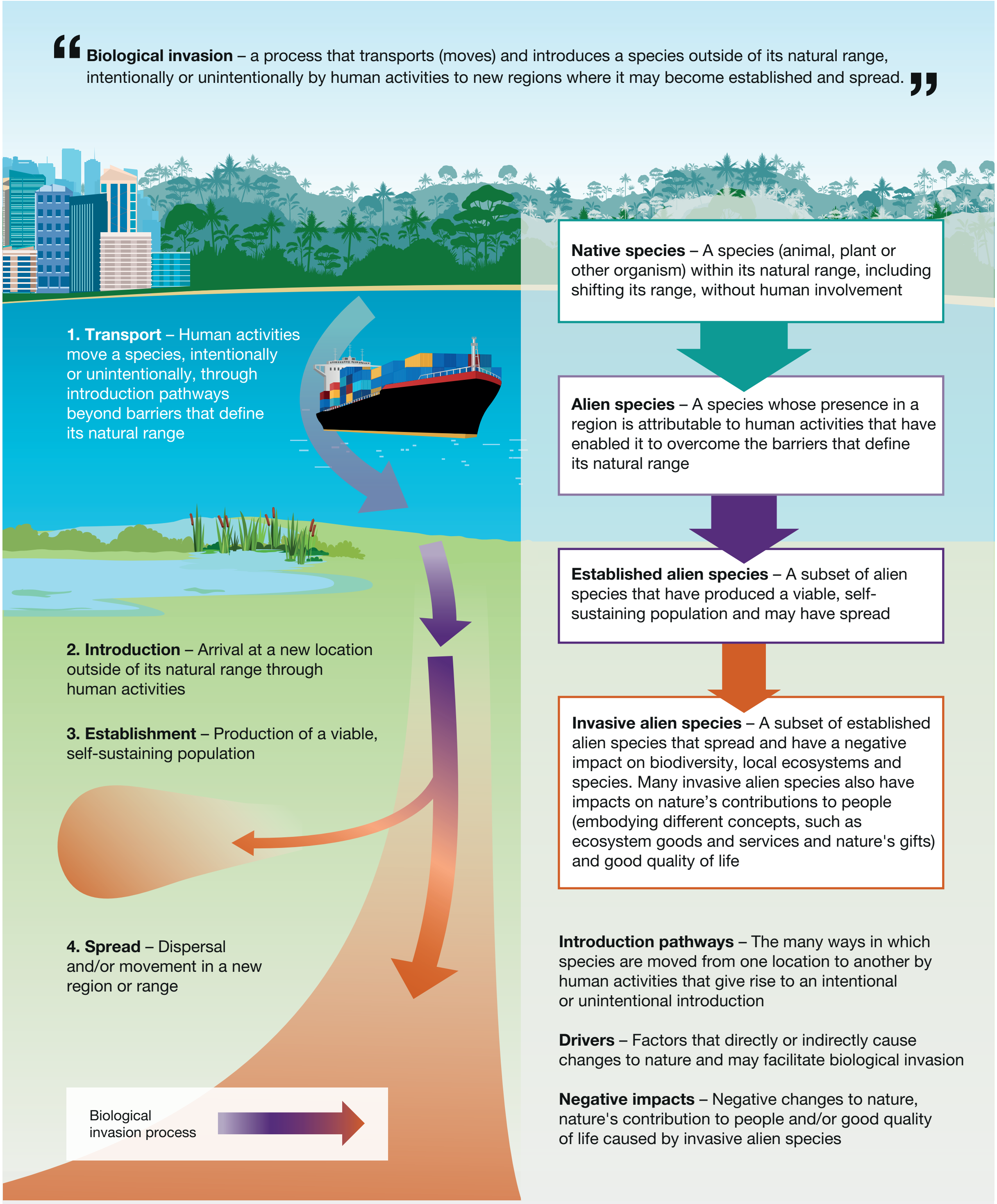

Figure SPM 1 **Key concepts within the biological invasion process.**[7]

Invasive alien species are one of the main direct drivers of change in nature. The biological invasion process comprises the following stages: transport, introduction, establishment and spread (or dispersal). Definitions of native, alien, established alien and invasive alien species are provided. Indirect and other direct drivers of change facilitate biological invasion.

7. This assessment acknowledges that national and local legislation to address biological invasions differ between countries and may include different definitions appropriate to specific national and local contexts.

KEY MESSAGES

KEY MESSAGES

A. Invasive alien species are a major threat to nature, nature's contributions to people, and good quality of life

Alien species are being introduced by human activities to all regions and biomes of the world at unprecedented rates. Some become invasive, causing negative and in some cases irreversible impacts on nature, including loss of uniqueness of biological communities, contributing to the unparalleled degree of deterioration of the biosphere upon which humanity depends.

KM-A1 People and nature are threatened by invasive alien species in all regions of Earth {A1} (Figure SPM.2). More than 37,000 established alien species have been introduced by human activities across all regions and biomes of Earth, with new alien species presently being recorded at an unprecedented rate of approximately 200 annually. Studies with evidence of negative impacts exist for more than 3,500 of these species, which are categorized as invasive alien species. The proportion of established alien species known to be invasive varies among taxonomic groups, ranging from 6 per cent of all alien plants to 22 per cent of all alien invertebrates. Twenty per cent of all impacts are reported from islands. A disproportionate number of documented negative impacts have been reported in terrestrial realms, especially in temperate and boreal forests and woodlands and cultivated areas (including agricultural land). About one quarter of documented negative impacts have been reported from aquatic realms, especially from inland surface waters/waterbodies and shelf ecosystems. In addition to their impacts on nature, about 16 per cent of invasive alien species have negative impacts on nature's contributions to people, and about 7 per cent on good quality of life.

KM-A2 Invasive alien species cause dramatic and, in some cases, irreversible changes to biodiversity and ecosystems, resulting in adverse and complex outcomes across all regions of Earth, including local and global species extinctions {A2, A3} (Figure SPM.3). Invasive alien species have contributed solely or alongside other drivers to 60 per cent of recorded global extinctions, and are the only driver in 16 per cent of the documented global animal and plant extinctions. Biotic homogenization, whereby biological communities around the world become more similar, is a major negative impact of invasive alien species, with consequences for the structure and functioning of ecosystems. Changes in the properties of ecosystems, such as soil and water characteristics, account for more than a quarter of documented impacts. The magnitude and types of impacts vary for different invasive alien species and across ecosystems and regions. The majority of documented global extinctions attributed mainly to invasive alien species have occurred on islands (90 per cent), and local extinctions account for 9 per cent of documented impacts of invasive alien species on islands. Some areas, despite being protected for nature conservation or being remote, are also vulnerable to the negative impacts of invasive alien species.

KM-A3 The economy, food security, water security and human health are profoundly and negatively affected by invasive alien species {A4, A5} (Figure SPM.3). In 2019, global annual costs of biological invasions were estimated to exceed US$423 billion. The vast majority of global costs (92 per cent) accrue from the negative impacts of invasive alien species on nature's contributions to people or on good quality of life, while only 8 per cent of that sum is related to management expenditures of biological invasions. The benefits to people that some invasive alien species provide do not mitigate or undo their negative impacts, which include harm to human health (such as disease transmission), livelihoods, water security and food security, with reduction in food supply being by far the most frequently reported impact (more than 66 per cent).

KM-A4 Invasive alien species can add to marginalization and inequity, including, in some contexts, gender- and age-differentiated impacts {A5, A6}. People with the greatest direct dependence on nature, including those involved in gender- and age-specific activities, such as fishing or weeding, may be disproportionately affected by invasive alien species. More than 2,300 invasive alien species are found on lands managed, used and/or owned by Indigenous Peoples across all regions of Earth, threatening their quality of life and often leading to general feelings of despair, sadness and stress. Indigenous Peoples and local communities, ethnic minorities, migrants, and poor rural and urban communities are disproportionately impacted by invasive alien vector-borne diseases. Biological invasions negatively affect the autonomy, rights and cultural identities of Indigenous Peoples and local communities through the loss of traditional livelihoods and knowledge, reduced mobility and access to land, and increased labour to manage the invasive alien species. Impact reports by some Indigenous Peoples and local communities document 92 per cent

negative impacts and 8 per cent positive impacts on nature caused by invasive alien species.

KM-A5 Overall, policies and their implementation have been insufficient in managing biological invasions and preventing and controlling invasive alien species {A7, A8}. Up to 2020, only partial progress was made towards international goals and targets (e.g., Aichi Biodiversity Target 9 and Sustainable Development Goal Target 15.8). While most countries have targets related to the management of biological invasions within their national biodiversity strategies and action plans, effective policies are often lacking or inadequately implemented. Eighty-three per cent of countries do not have national legislation or regulations directed specifically toward the prevention and control of invasive alien species. Policy relevant to biological invasions is also fragmented within countries and across sectors. To date, capacity to respond to biological invasions has varied widely across regions, with nearly half of all countries (45 per cent) not investing in management of invasive alien species (SDG indicator 15.8.1). Differences in perception, including conflicting interests and values, of the importance and urgency of the threat of invasive alien species, coupled with lack of awareness of the need for a collective and coordinated response, as well as gaps in data and knowledge, can hinder the management of invasive alien species. Economic development policies and those aiming to manage other drivers of change sometimes facilitate biological invasions. Demographic drivers also facilitate the introduction and spread of invasive alien species while acknowledging that drivers differ across regions and level of impact. The lack of border biosecurity (such as inspections undertaken by quarantine officers of commodities, goods and people) in one country weakens the efficacy of such measures in other countries.

B. Globally, invasive alien species and their impacts are increasing rapidly and are predicted to continue rising in the future

The threats from invasive alien species are increasing in all regions of Earth and are predicted to do so in the future. Even without the introduction of new species, existing populations of invasive alien species will continue spreading through all ecosystems. Amplification of and interactions among direct and indirect drivers of change will profoundly shape and exacerbate the future threats from invasive alien species.

KM-B1 Many human activities facilitate the transport, introduction, establishment and spread of invasive alien species {B9, B11, B12, B14}

(Figure SPM.5). Many invasive alien species have been intentionally introduced outside their natural range around the world for their perceived benefits without consideration or knowledge of their negative impacts, but there have also been many unintentional introductions, including as contaminants of traded goods and stowaways in shipments. Indirect drivers of change, particularly those associated with economic activities, of which international trade is the most important, are increasingly facilitating transport and introduction, the early stages of biological invasion. Direct drivers, particularly land- and sea-use change and climate change, are increasingly important later in the biological invasion process, facilitating the establishment and spread of invasive alien species, with fragmented ecosystems being more vulnerable to invasive alien species. Transport and utility infrastructures in terrestrial and aquatic environments can create corridors that facilitate the spread of invasive alien species, including into remote, undisturbed and protected areas. For some invasive alien species, the spread is immediate, but others only begin to spread long after first introduction, meaning that currently observed threats of invasive alien species can lead to underestimation of the magnitude of the future impact. Invasive alien species may increase in numbers after a long period at low density as a result of changes in interactions with other species, for example as a result of the introduction of a missing dispersal agent or the removal of a competitor.

KM-B2 The threats from invasive alien species are increasing markedly in all regions of Earth, with the current unparalleled high rate of introductions predicted to rise even higher in the future {B10} (Figure SPM.4). The number of alien species has been rising continuously for centuries in all regions, and the global economic costs of invasive alien species have quadrupled every decade since 1970. Even without the introduction of new species, already established alien species given the opportunity, may continue to expand their geographic ranges into new countries, regions and ecosystems, including remote environments. Under a “business-as-usual” scenario, which assumes that trends of drivers will continue as observed in the past, by 2050 the total number of alien species globally is expected to be about one-third higher than in 2005. However, the number of alien species worldwide is expected to increase faster than predicted under the business-as-usual scenario.

KM-B3 The ongoing amplification of drivers of change in nature may substantially increase the number of invasive alien species and their impacts in the future {B9, B11, B12, B14}. The causal links between indirect and direct drivers imply that ongoing and future amplification of these drivers will increase the frequency and extent of biological invasions and the impacts of invasive alien species, which, in some cases, may exacerbate the impacts of other drivers. At a global scale, the number of invasive alien species and their negative impacts are likely to increase due to the amplification of multiple drivers including but not limited to demographic, economic and land-use and sea-use change while noting regional variation. Additionally, climate change will further exacerbate the establishment of some invasive alien species and will be a major cause of future establishment and spread. Delays in the response of invasive alien species to drivers of change may result in a long legacy of future biological invasions due to past and present amplification of drivers.

KM-B4 The magnitude of the future threat from invasive alien species is difficult to predict because of complex interactions and feedback among direct and indirect drivers of change in nature {B10, B13, B14}. Climate change interacting with land- and sea-use change is predicted to profoundly shape and amplify the future threat from invasive alien species. Interactions among climate change, land-use change and invasive alien species can alter and intensify natural disturbance regimes, such as wildfires. Variations in human perceptions and values add yet another level of complexity, as sociocultural drivers interact with other indirect drivers and influence direct drivers. Such interactions may lead to unprecedented numbers of invasive alien species, with the consequent amplification of their impacts.

C. Invasive alien species and their negative impacts can be prevented and mitigated through effective management

Curbing the rising number of invasive alien species and reducing their spread and impacts are achievable through management actions in the short as well as long term. There are many decision frameworks and approaches for supporting management of invasive alien species at all stages of the biological invasion process. Prevention is the best option, but early detection, eradication, containment and control are also effective in specific contexts. Management of biological invasions benefits from engagement with stakeholders and Indigenous Peoples and local communities.

KM-C1 The number and impacts of invasive alien species can be reduced through management of biological invasions {C15, C16, C17, C18, C22, C23} (Figure SPM.6, Table SPM.1). There are decision-

making frameworks and tools for inclusively identifying and supporting management goals related to (a) management of pathways of introduction and spread of invasive alien species; (b) management of target invasive alien species at either local or landscape scales; and (c) site-based or ecosystem-based management. There are many sources of accessible literature and information, tools, and novel and emerging technologies, including biotechnology, bioinformatics, eDNA, remote sensing and data analytics, for supporting the management of biological invasions. Consideration of both potential benefits and risks of the management of biological invasions can improve outcomes. A risk assessment and a risk management framework in line with a precautionary approach, as appropriate, can be effective to guide management actions, including the use of novel and emerging and environmentally sound technologies. The success of any management programme depends on the availability of adequate, sustained resources, including for building capacity, which is sometimes lacking, especially in some developing countries. Multi-stakeholder engagement, including risk communication and context-specific application, can improve public acceptability and adoption of new tools and technologies for managing biological invasions.

KM-C2 Prevention and preparedness are the most cost-effective options and thus crucial for managing the threats from invasive alien species {C15, C17, C18}. Prevention can be achieved through pathway management, including strictly enforced import controls, pre-border, border and post-border biosecurity, and measures to address escape from confinement. Prevention is particularly critical in marine and connected water systems, where most attempts at eradicating or containing invasive alien species have mostly failed. Prevention has been particularly effective on islands. Preparedness includes border surveillance, early detection and rapid response planning, and is critical to reduce rates of establishment. Horizon scanning and risk analysis can support prevention and preparedness by prioritizing emerging invasive alien species. Sustained and adequate funding, capacity-building, technical and scientific cooperation, transfer of technology, monitoring, relevant and appropriate biosecurity legislation and enforcement, and quarantine and inspection facilities are necessary for effective prevention measures.

KM-C3 Eradication has been successful, especially for small and slow-spreading populations of invasive alien species in isolated ecosystems {C19}. Over the last 100 years, 88 per cent of eradication attempts on 998 islands have proven successful, especially for invasive alien vertebrates. Large-scale eradications have been achieved but in many cases are likely to be infeasible. There are also examples of eradication of invasive alien plants and invertebrates, particularly for those with limited distribution. Adoption of appropriate tools and technologies and involvement of relevant stakeholders underpin and improve the success of eradication programmes. Sustained investment is required for eradication programmes but they are generally more cost-effective than long term and permanent control or the costs incurred through inaction.

KM-C4 Containment and control can be an effective option for invasive alien species that cannot be eradicated for various reasons from terrestrial and closed water systems, but most attempts in marine and connected water systems have been largely ineffective {C20}. Physical control alongside chemical control options in terrestrial and closed water systems are generally only effective at a local scale and can have non-target effects. Biological control can be applied for widely distributed invasive alien species and has been successful in managing some invasive alien plants, invertebrates and, to a lesser extent, plant pathogenic microbes and vertebrates, but it may also have non-target effects if not well regulated. International standards and risk-based regulatory frameworks for biological control have been used in many countries to manage risks, and continue to be successfully applied. Integrated management, where more than one containment or control option are used, can improve outcomes.

KM-C5 The recovery of ecosystem functions and nature's contributions to people can be achieved through adaptive management, including ecosystem restoration in terrestrial and closed water systems {C21}. Management outcomes can be improved by the integration of site- and/or ecosystem-based management options that enhance ecosystem function and resilience. Frequent long-term monitoring of sites ensures early detection of invasive alien species, including re-invasions, and can inform further management actions. In marine and connected water systems, ecosystem restoration has so far proved to be largely ineffective. Adaptive management, possibly combining multiple options, will improve management of biological invasions under ongoing climate and land-use change. Integrating site and/or ecosystem-based approaches can improve management outcomes of biological invasions and also enhance ecosystem functioning under ongoing climate and land-use change.

KM-C6 Engagement and collaboration with stakeholders and Indigenous Peoples and local communities improve outcomes of management actions for biological invasions {C23, C24}. Engaging stakeholders, including the private sector, and Indigenous Peoples and local communities in the collaborative management of biological invasions is important for social acceptability and improving environmental, social and economic outcomes, particularly where there are conflicting perceptions of the value of invasive alien species and the ethics of management options. Management actions also benefit from sharing and collaboration across knowledge systems. Recognizing Indigenous Peoples' and local communities' knowledge, rights and customary governance systems in accordance with national legislation also helps to improve long-term management.

D. Ambitious progress to manage biological invasions[8] can be achieved with integrated governance

One of the greatest threats to biodiversity, invasive alien species can be overcome through a context-specific integrated governance approach to biological invasions, including well-resourced, coordinated and sustained strategic actions, with closer collaboration across sectors and countries. Managing biological invasions is realistic and achievable, with substantial benefits for nature and people.

KM-D1 Through a complementary set of strategic actions, integrated governance can limit the global problem of invasive alien species throughout the biological invasion process and at local, national and regional scales {D25}. Strategic actions to prevent the introduction and impact of invasive alien species include: enhancing coordination and collaboration across international and regional mechanisms; developing and adopting effective and achievable national strategies; sharing efforts and commitment and understanding the specific role of all actors; improving policy coherence; broad engagement across all stakeholders and Indigenous Peoples and local communities; resourcing innovation, research and technology; and supporting information systems, infrastructures and data sharing.

KM-D2 The threat of invasive alien species could be reduced with closer collaboration and coordination across sectors and countries to support the management of biological invasions {D26, D30} (Figure SPM.7). International, national and local agencies involved in developing policies for the environment, agriculture, aquaculture, fishing, forestry, horticulture, border control, shipping (including biofouling), tourism, trade (including online trade in animals, plants, and other organisms), community and regional development (including infrastructure), transportation and the health sector can all play a role in developing a coherent approach

8. This assessment acknowledges that national and local legislation to address biological invasions differ between countries and may include different definitions appropriate to specific national and local contexts.

to managing biological invasions and preventing and controlling invasive alien species. Enhancing coordination and collaboration across international and regional mechanisms is one of the key strategic actions for rapid and transformative progress. International and regional partnerships can improve management of biological invasions. Collaboration and co-development with Indigenous Peoples and local communities can increase the effectiveness of implemented strategies.

KM-D3 The Kunming-Montreal Global Biodiversity Framework provides an opportunity for national governments to develop or update aspirational, ambitious and realistic approaches to prevent and control invasive alien species {D27, D28} (Figure SPM.7). Implementation-focused national biodiversity strategies and action plans can help to spur strategic actions and establish the properties of the governance systems required for the successful prevention and control of invasive alien species and the management of biological invasions, and work towards delivering Target 6. Coordinated efforts to strengthen national regulatory instruments are also priorities, including those for online trading and the creation of appropriate policies for the development and use of environmentally sound technologies, as well as making available data and information accessible. Market-based instruments such as tax relief and subsidization can be used to incentivize action and spur relevant investment. Sharing efforts and commitment, understanding the specific roles of all actors and encouraging engagement across sectors on prevention, control and environmental liability are integral to the effective management of biological invasions.

KM-D4 Preventing and controlling invasive alien species can strengthen the effectiveness of policies designed to respond to other threats to biodiversity and contribute to achieving several Sustainable Development Goals {D26, D33}. Awareness of the risks of biological invasions will contribute to the effective delivery of several of the Sustainable Development Goals, especially those addressing the conservation of marine biodiversity (Goal 14) and terrestrial biodiversity (Goal 15, including but not restricted to Target 15.8), food security (Goal 2), sustainable economic growth (Goal 8) and sustainable cities (Goal 11), as well as climate change (Goal 13) and health and wellbeing (Goal 3). Existing collaborative and multisectoral approaches (e.g., One Health) could provide frameworks for cross-disciplinary thinking and could contribute to the management of biological invasions.

KM-D5 Open and interoperable information systems will improve the coordination and effectiveness of the management of biological invasions, within and across countries {D31, D32}. By delivering current data to relevant actors, information systems can facilitate the prioritization of actions and allow for early detection and rapid response. Information systems can also support improved governance and help develop indicators of biological invasions, which in turn feed into policy support tools. Collaboration between biological invasion experts and across knowledge systems in all regions, and enhancement of research capacity where needed, can improve data and information availability and the understanding of the context-specific features of biological invasions and their impacts.

KM-D6 Public awareness, commitment and engagement, and capacity-building, are crucial for the prevention and control of invasive alien species {D29, D31, D32} (Table SPM.2). Advances can be achieved through adequately and sustainably resourced public awareness campaigns, education, citizen science, and targeted investment in research innovation and environmentally sound technology. Public engagement with citizen science platforms and community-driven eradication campaigns can raise awareness and contribute to actions that reduce the threat of invasive alien species. This can also be aligned with efforts to share efforts and commitment and understand the specific roles of all actors. Communication strategies based on evidence can help to bring about community action on biological invasions by supporting the co-design of management actions, knowledge exchange and enhanced partnerships among stakeholders.

KM-D7 There is compelling evidence for immediate and sustained action to manage biological invasions and mitigate the negative impacts of invasive alien species {D32, D33} (Table SPM.2). With sufficient resources, political will and long-term commitment, preventing and controlling invasive alien species are attainable goals that will yield significant long-term benefits for people and nature. Increasing the availability and accessibility of information and means of implementation and addressing major knowledge gaps on biological invasions, particularly in developing countries, would result in more robust and effective policy instruments and management actions. Additional efforts and cooperation are particularly needed to improve data collection in Africa, Latin America and the Caribbean and Asia.

BACK-GROUND

BACKGROUND

A. Invasive alien species are a major threat to nature, nature's contributions to people, and good quality of life

A1 More than 37,000 established alien species, including more than 3,500 invasive alien species with documented impacts, have been recorded worldwide *(well established)* {2.1.4, 4.2}. Alien species (plants, animals, fungi and microorganisms, including pathogens) are being introduced globally at an unprecedented rate; currently, approximately 200 new alien species are recorded every year *(well established)* {2.2.1}. Invasive alien species represent a subset of alien species, consisting of those that have established and spread and are known to have a negative impact on nature and, in some cases, people **(Figure SPM.1)**. Although their numbers are likely to be underestimated and expected to increase, to date 1,061 alien plants (6 per cent of all established alien plants), 1,852 alien invertebrates (22 per cent), 461 alien vertebrates (14 per cent) and 141 alien microbes (11 per cent) are known to be invasive globally *(established but incomplete)* {4.2}. Although some invasive alien species can provide benefits for people (e.g., through provision of food and fibre), those benefits do not mitigate or undo their negative impacts on nature, nature's contributions to people, and good quality of life across all regions and taxa globally *(well established)* {1.3.4, 4.1.2, 4.3, 4.4, 4.5}. In addition to their impacts on nature, about 16 per cent of invasive alien species have negative impacts on nature's contributions to people, and about 7 per cent on good quality of life **(Figure SPM.2)** *(established but incomplete)* {4.2}. Based on data and information included in this assessment, most impacts are reported in the Americas (34 per cent), Europe and Central Asia (31 per cent) and Asia-Pacific (25 per cent), with fewer reported in Africa (7 per cent) *(established but incomplete)* {4.2}. Twenty per cent of all impacts are reported from islands *(established but incomplete)* {4.2}. A disproportionate number of documented negative impacts have been reported from the terrestrial realm (75 per cent), especially temperate and boreal forests and woodlands and cultivated areas (including agricultural land) *(established but incomplete)* {Table 4.2}. About one quarter of the documented negative impacts have been reported from aquatic realms (freshwater: 14 per cent; marine: 10 per cent), especially from inland surface waters/waterbodies and shelf ecosystems *(established but incomplete)* {Table 4.2}.

A2 Invasive alien species are a major direct driver of change, causing biodiversity loss, including local and global species extinctions (Figures SPM.2 and 3) *(well established)* {4.3.1}. Invasive alien species have contributed solely or alongside other drivers of change to 60 per cent of recorded global animal and plant extinctions *(established but incomplete)* {Box 4.4, 4.3.1}, while invasive alien species are the only driver attributed to 16 per cent of documented global extinctions *(established but incomplete)* {Box 4.4}. The majority of documented global extinctions (90 per cent) with invasive alien species as one of the major causes are reported from islands *(established but incomplete)* {Box 4.4}. At least 218 invasive alien species have caused 1,215 documented local extinctions of native species across all taxa **(Figure SPM.3)** *(established but incomplete)* {4.3.1}. Invasive alien species harm native species most often by changing ecosystem properties (27 per cent), for example soil and water characteristics, and through competition between species (24 per cent), predation (18 per cent) and herbivory (12 per cent) *(established but incomplete)* {4.3.1.3}. The majority of reports of the impacts of invasive alien species on native species document negative effects (85 per cent), primarily negatively impacting the growth, survival and reproduction of individuals, which lead to local population declines and local and global extinctions *(well established)* {4.3.1}. Some invasive alien species have a profound ecological impact that spans various levels, from individual species and communities to whole ecosystems, resulting in complex, undesirable and in some cases irreversible outcomes when the system has crossed a threshold beyond which ecosystem restoration is not possible *(well established)* {Box 1.5, Box 4.12, 4.3.3}. For example, *Castor canadensis* (North American beaver) and *Magallana gigas* (Pacific oyster) change ecosystem properties by transforming habitats, with cascading effects on a myriad of native species *(well established)* {4.3.2.1, Box 4.11}. On Christmas Island, the arrival of the invasive alien *Anoplolepis gracilipes* (yellow crazy ant) caused the decline of the native Christmas Island *Gecarcoidea natalis* (red crabs), which resulted in the population explosion of the invasive alien *Lissachatina fulica* (giant African land snail) *(well established)* {3.3.5.1}. Increased biotic homogenization (or loss of uniqueness) of biological communities is a major negative impact of invasive alien species *(well established)* {1.3.4}. The magnitude of the negative impacts of invasive alien species on nature depends on the context, and the factors that determine the highest magnitudes of impact are not well understood *(established but incomplete)* {Box 4.9, 4.3.2.1, 4.7.1}. For

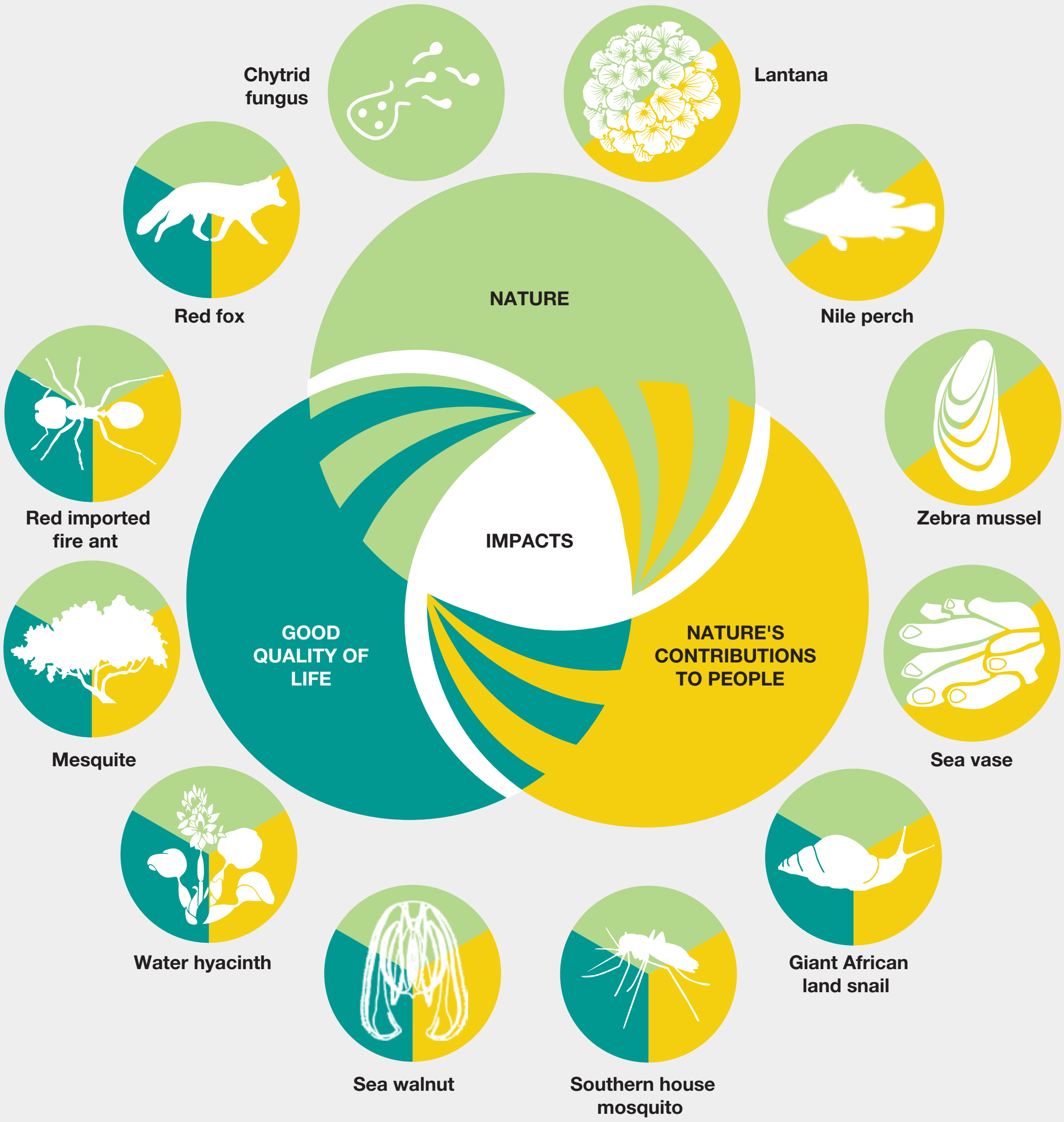

Figure SPM 2 **Examples of invasive alien species with a negative impact on nature (green), and, in some cases, nature's contributions to people (yellow) and/or good quality of life (teal).**

Many invasive alien species have documented negative cross-cutting impacts, indicated by multiple colours in the examples: 16 per cent of invasive alien species have a negative impact on both nature and nature's contributions to people; 7 per cent on both nature and good quality of life; and 5 per cent on nature, nature's contributions to people and good quality of life {4.2}. The scientific names of the example species are *Lantana camara* (lantana); *Lates niloticus* (Nile perch); *Dreissena polymorpha* (zebra mussel); *Ciona intestinalis* (sea vase); *Lissachatina fulica* (giant African land snail); *Culex quinquefasciatus* (southern house mosquito); *Mnemiopsis leidyi* (sea walnut); *Pontederia crassipes* (water hyacinth); *Prosopis juliflora* (mesquite); *Solenopsis invicta* (red imported fire ant); *Vulpes vulpes* (red fox); and *Batrachochytrium dendrobatidis* (chytrid fungus).

example, the ctenophore *Mnemiopsis leidyi* (sea walnut) has depleted zooplankton, the main food source of the anchovy, and consequently contributed to the collapse of anchovy populations in the Black Sea, but this has not occurred in the Mediterranean Sea, the Baltic Sea or the North Sea *(well established)* {4.3.2.3}.

A3 On islands, invasive alien species are a major cause of biodiversity loss *(well established)* {Box 2.5, 4.3.1.1, Box 4.4}. Islands, and particularly remote islands with high endemism, are more susceptible to impacts from invasive alien species than mainlands *(well established)* {1.6.8, 4.3.1.1}. Indeed, in addition to the majority of documented global extinctions attributed mainly to invasive alien species occurring on islands, local extinctions account for 9 per cent of documented impacts of invasive alien species on islands, in contrast to 4 per cent on mainlands *(well established)* {4.3.1.1}. For example, *Boiga irregularis* (brown tree snake) caused the global extinction of *Myiagra freycineti* (Guam flycatcher) and local extinction or serious population reduction for many other resident bird species in Guam *(well established)* {4.3.1}. Islands are also vulnerable to climate change, which can increase the rate of establishment and spread of many invasive alien species *(well established)* {Box 2.5}. Many invasive alien species on islands only occupy a small portion of their predicted range and are likely to expand further *(established but incomplete)* {Box 2.5}. The number of alien plants exceeds the total number of native plants on more than one quarter of islands *(well established)* {Box 2.5}. Invasive alien species have been reported in areas protected for nature conservation, some remote areas (e.g., high mountains), and also in tundra and deserts, which emphasizes that these areas, despite being protected for nature conservation or remote, are also vulnerable to the negative impacts of invasive alien species *(well established)* {Box 2.4, 4.3.1.2, 4.3.2.1}. Fifty-three invasive alien species have caused the local extinctions of 240 native species in protected areas globally *(established but incomplete)* {4.3.1.2}. The invasive alien *Rattus rattus* (black rat) has been documented as the only cause of the global extinction of *Nesoryzomys darwini* and *Nesoryzomys indefessus* (rice rats), which were endemic to the protected areas of the Galapagos Islands *(well established)* {4.3.1}.

A4 Invasive alien species adversely affect the full range of nature's contributions to people, imposing an economic burden *(well established)* {4.4.1}. Some alien species have been intentionally introduced for their benefits to people, often without consideration or knowledge of their negative impacts *(well established)* {3.3.1}. However, nearly 80 per cent of the documented impacts of invasive alien species on nature's contributions to people are negative *(well established)* {4.4.1}. Reduction in food supply is by far the most frequently reported impact across all taxa and regions *(well established)* {4.4.1, 4.6.2}. In terrestrial systems, invasive alien plants are the taxonomic group most frequently reported as having a negative impact, particularly in cultivated areas and temperate and boreal forests *(well established)* {4.4.2.1}. For example, in north-western Europe, *Picea sitchensis* (Sitka spruce) severely alters habitats such as coastal heathlands and mires, which are important habitats for threatened and endangered plants, birds and other species, and for local cultural heritage *(well established)* {4.3.2.1}. In coastal areas, invasive alien invertebrates are the most frequently reported taxonomic group with an impact on nature's contributions to people, particularly provision of food *(well established)* {4.4.2.3}. For example, *Carcinus maenas* (European shore crab) has had an impact on commercial shellfish beds in New England and Canada, *Asterias amurensis* (northern Pacific seastar) and *Ciona intestinalis* (sea vase) have negatively affected mariculture and fisheries along the Korean coast, and *Mytilopsis sallei* (Caribbean false mussel) has displaced native clams and oysters that are locally important fishery resources in India *(well established)* {4.4.2.3}. In 2019, global annual costs of biological invasions were estimated to exceed US$423 billion, with variations across regions, but this is likely to be a gross underestimate **(Figure SPM.3)** *(established but incomplete)* {Box 4.13}. Ninety-two per cent of this cost is attributed to the damage that the invasive alien species have caused to nature's contributions to people and good quality of life; only 8 per cent is related to the management expenditures for biological invasions *(established but incomplete)* {Box 4.13}. Economic benefits are often gained by a few people or sectors while costs, often long-term ones, are borne by many others *(established but incomplete)* {3.2.3.5, 4.2.1, 6.2.2(6)}.

A5 Invasive alien species overwhelmingly undermine good quality of life *(established but incomplete)* {4.5, 4.6.3}. Invasive alien species can threaten livelihoods, water and food security, economies and human health (e.g., causing diseases, allergies and physical injuries) **(Figure SPM.3)** *(well established)* {4.5.1, 4.5.1.3}, with 85 per cent of the documented impacts of invasive alien species on good quality of life being negative **(Figure SPM.3)** *(well established)* {4.5.1}. Invasive alien species can also serve as vectors for infectious zoonotic diseases that can lead to epidemics, such as malaria, dengue fever, chikungunya, Zika, yellow fever and West Nile fever, which are transmitted by invasive mosquito species (e.g., *Aedes albopictus* and *Aedes aegypti*) *(well established)* {Box 1.14, 4.5.1.3}. Invasive alien plants can impact human health directly, particularly through the production of highly allergenic pollen, for example, *Prosopis juliflora* (mesquite) and *Ambrosia artemisiifolia* (common ragweed) *(well established)* {4.5.1.3}. Indigenous Peoples and local communities, ethnic minorities, migrants, poor rural and urban communities are disproportionately impacted by invasive alien vector-borne diseases *(established but incomplete)* {4.5.1}. Although there is limited research on the interplay between gender relations and invasive alien species *(established but incomplete)* {4.5.1, 4.7.2}, there is some evidence of inequities and marginalization in gender- and age-specific activities where invasive alien species impede access to natural resources or require management *(established but incomplete)* {4.5.1, 5.2, 5.2.1, 5.5.5}. For example, in Lake Victoria artisanal fisheries mainly involving men have declined following the introduction, establishment and spread of the invasive alien plant *Pontederia crassipes* (water hyacinth), which has led to the depletion of tilapia *(established but incomplete)*

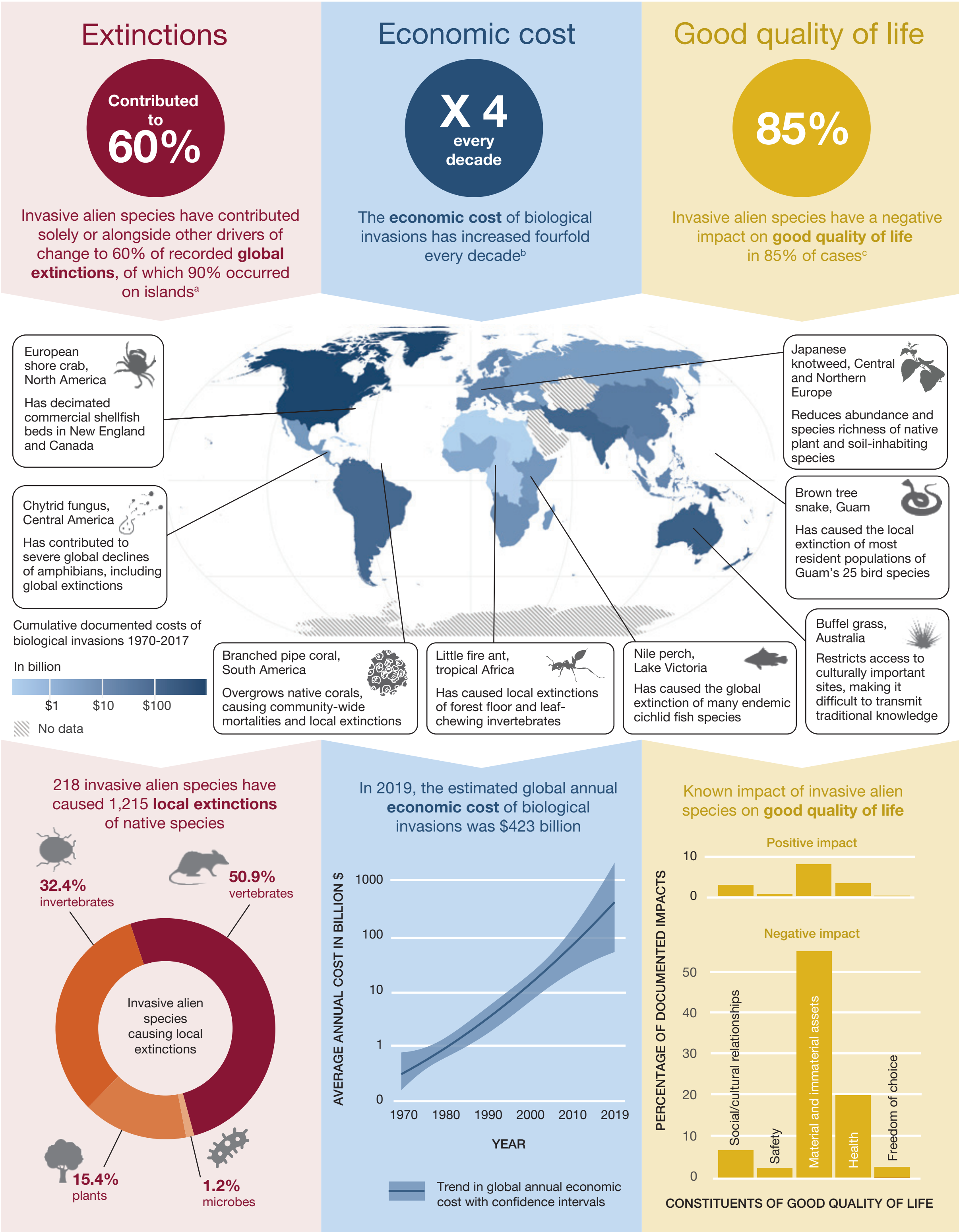

Figure SPM 3 **Extent of the problems caused by invasive alien species.**

Illustrative examples of the impacts of invasive alien species on native species (red; left column), on the economy (blue; centre column) and on good quality of life (yellow; right column). The top row illustrates the documented numbers of global and local extinctions of native species to which invasive alien species have contributed (left); the rate of increase in the economic cost of biological invasions per decade (centre); and the percentage of cases where the impact of invasive alien species on good quality of life is reported as negative (right). The map in the centre row shows the documented cumulative economic cost of invasive alien species per IPBES subregion from 1970 to 2017. The case studies illustrate a variety of impacts of invasive alien species on both nature and good quality of life in different geographic regions, taxonomic groups and realms, but are not meant to be representative. The bottom row shows the taxonomic distribution (i.e., plants, invertebrates, vertebrates and microbes, including fungi) of the percentage of invasive alien species documented as causing local extinctions of native species (left); the estimated global annual average economic cost of biological invasions in billions of United States dollars (centre); and the percentage of the number of documented positive and negative impacts of invasive alien species on the constituents of good quality of life (i.e., freedom of choice, health, material and immaterial assets, safety, social and cultural relationships) (right). a: {4.3.1, Table 4.3}; b: {4.4.1, Box 4.13}; c: {4.5.1, Table 4.20}. The scientific names of the example species are *Carcinus maenas* (European shore crab); *Batrachochytrium dendrobatidis* (chytrid fungus); *Carijoa riisei* (branched pipe coral); *Wasmannia auropunctata* (little fire ant); *Lates niloticus* (Nile perch); *Cenchrus ciliaris* (buffel grass); *Boiga irregularis* (brown tree snake); and *Reynoutria japonica* (Japanese knotweed).

{4.5.1}. In East Africa, management of the invasive alien plant *Opuntia* spp. (prickly pear) requires repeated weeding by hand, which is often undertaken by women and children and has in many cases become their most time-consuming activity *(established but incomplete)* {5.5.5}. Invasive alien species may be introduced for economic development, for example through financing large-scale infrastructures *(well established)* {3.2.5, 3.3.1.3, 3.3.1.4, Box 3.11, 3.3.1.1, 3.3.2.1.1}. In some cases, invasive alien species have been unintentionally transported and introduced through emergency relief and aid (e.g., seeds of the invasive alien plant *Parthenium hysterophorus* (parthenium weed) arrived with grain in aid shipments in several countries) *(well established)* {3.2.2.3}, increasing the risk of possible negative impacts on quality of life *(established but incomplete)* {4.5.1, 4.6.3}.

A6 Many invasive alien species have been documented on lands managed, used and/or owned by Indigenous Peoples and local communities *(established but incomplete)* {Box 2.6; 4.6}. More than 2,300 invasive alien species have been documented on lands managed, used and/or owned by Indigenous Peoples, with some negatively affecting their quality of life and cultural identities. Indigenous lands in Oceania and North America have particularly high numbers of recorded invasive alien species *(established but incomplete)* {Box 2.6}. However, numbers of invasive alien species are, on average, consistently lower on Indigenous lands compared to other lands *(established but incomplete)* {Box 2.6}. Many Indigenous Peoples and local communities emphasize the inter-relatedness of the land, water and humans and other species, which can lead to a range of diverse perceptions of specific invasive alien species *(well established)* {1.6.7.1}. In some cases, Indigenous Peoples and local communities may consider an invasive alien species a valued part of their nature *(established but incomplete)* {1.6.7.1}. There are also examples where Indigenous Peoples and local communities have created new income sources by relying on invasive alien species *(well established)* {4.5.1, 4.6.2}, but that often occurs through necessity rather than choice. However, impact reports by some Indigenous Peoples and local communities document 68 per cent negative impacts and 32 per cent positive impacts on their good quality of life caused by invasive alien species *(established but incomplete)* {4.6.1, 4.6.3.2, Table 4.33}. Indigenous Peoples and local communities often have a good understanding of how the complex interactions among drivers facilitate the introduction and spread of invasive alien species on their lands *(established but incomplete)* {3.2.3.6, Box 3.15}. For example, Indigenous Peoples and local communities recognize that the use of invasive alien species for food, fibre, income generation or medicinal purposes can cause negative impacts on nature's contributions to people and their good quality of life *(well established)* {3.2.3.6, Box 3.6}, especially in situations where the native species they traditionally depended on for those benefits have declined *(established but incomplete)* {3.2.3.6; 3.2.5}. Impact reports by some Indigenous Peoples and local communities document 92 per cent negative impacts and 8 per cent positive impacts on nature caused by invasive alien species *(established but incomplete)* {Table 4.31}. Negative impact reports include water security and human and livestock health, as well as acknowledging that invasive alien species limit access to traditional lands, reduce mobility and require increased labour to manage *(established but incomplete)* {Box 4.9, 4.5.1, 4.5.1.4, 4.6.3.1, 4.6.3.2, 5.5.5}. Invasive alien species can also adversely affect the autonomy, rights and cultural identity of Indigenous Peoples and local communities *(established but incomplete)* {Box 4.15} through the loss of traditional livelihoods, knowledge and cultural practices *(well established)* {4.6.3.2}, often leading to general feelings of despair, sadness and stress *(established but incomplete)* {4.6.3.2}.

A7 Perceptions of the threat of invasive alien species can vary depending on different human perspectives *(well established)* {1.5.2}. Perceptions of specific invasive alien species and their value differ among and within stakeholder groups and Indigenous Peoples and local communities, as different community members can experience different impacts depending on gender, age, livelihood and a multitude of other factors *(established but incomplete)* {1.5.2, 1.6.7.1, 3.2.1, 5.6.1.2}. Value conflicts arise when invasive alien species are considered to be a major threat by some stakeholders and beneficial by others *(well established)* {5.6.1.2}. An invasive alien species may

Box SPM 1 **Voluntary codes of conduct can complement legislation for managing the risks of transport and the introduction of invasive alien species through trade.**

Voluntary codes of conduct have limits, but they provide practical and concise guidance in establishing common standards of good practice and sustainable attitudes and behaviours for managing the risks of transport and the introduction of invasive alien species through trade. For example, awareness of horticulture as a major pathway for the introduction of many (46 per cent) invasive alien plants worldwide {3.2.3.2} has led to industry–government collaboration that has resulted in the implementation of voluntary codes of conduct for the horticultural industry, complementing legislation to ban the sales of invasive alien plants considered to be high risk {Box 6.6}. When designed in a collaborative manner, codes of conduct can help producers and consumers make informed choices. The adoption of voluntary codes of conduct can encourage e-commerce platforms to adopt better practices by screening their lists for invasive alien species, complying with relevant legislation and providing information on the species, including taxonomy, potential invasiveness and appropriate measures that a buyer could use to prevent escape. Codes of conduct have also been developed in Europe for other activities that can facilitate the introduction of invasive alien species, including boating, botanic gardens, horticulture, hunting, international travel, plantation forestry, pets, protected areas, e-commerce, recreational fishing, zoological gardens and aquaria.

Published in 2013 by the Council of Europe, the *European Code of Conduct for Botanic Gardens on Invasive Alien Species* outlines voluntary principles for all botanic garden personnel to support them in protecting ecosystems from the impacts of invasive alien species.

See: Heywood, V. H., & Sharrock, S. (2013). *European Code of Conduct for Botanic Gardens on Invasive Alien Species.* Council of Europe Publishing, F-67075 Strasbourg www.coe.int/Biodiversity

have been intentionally introduced for a particular purpose, including to mitigate other drivers of change *(well established)* {Box 3.9}, but can have negative impacts on other sectors *(well established)* {3.3.1.1, 3.2.5, 5.6.1.2}. For example, introduced pigs are important culturally in Hawaii and are hunted for subsistence, ceremony and recreation, despite causing severe negative impacts by driving and maintaining the spread of invasive alien plants within Hawaiian rainforest *(established but incomplete)* {5.6.1.2}. Divergence of perceptions of invasive alien species can prevent effective decision-making and management *(established but incomplete)* {5.6.1.2, 6.2.2(9)}. The management of invasive alien species can, in some cases, raise multiple ethical debates about animal welfare and rights *(well established)* {1.5.3, 5.6.2.1, Box 6.13} (e.g., the challenges of effectively managing the biological invasion of *Hippopotamus amphibius* (African hippopotamus) in Colombia due to it being considered a charismatic species *(established but incomplete)* {5.4.3.1}).

A8 Current policy instruments for biological invasions have led to only partial progress towards international Targets on invasive alien species, including Aichi Biodiversity Target 9 and Sustainable Development Goal Target 15.8 *(well established)* {6.1.2, 6.1.3}. Most countries (80 per cent, 156 out of 196) have targets for the management of biological invasions within their national biodiversity strategies and action plans, 74 per cent (145) of which are aligned with Aichi Biodiversity Target 9 *(well established)* {6.1.2}. Assessment of the progress towards meeting Aichi Biodiversity Target 9 concluded that there was still a considerable gap between the development and adoption of invasive alien species policy and implementation at national levels *(well established)* {6.1.2}. Although the number of countries with national invasive alien species checklists, including databases, has more than doubled in the last decade (196 countries in 2022) **(Table SPM.A3)** {6.1.3}, 83 per cent do not have national legislation or regulations specifically on invasive alien species *(well established)* {6.1.3}, which also increases the risk of biological invasions for neighbouring countries *(well established)* {6.3.2.1}. Only 17 per cent of countries have national legislation for biological invasions, whereas an estimated 69 per cent have biological invasions-specific legislation as part of legislation in other sectors *(well established)* {6.1.2, 6.1.3}. Although many agribusinesses do not manage the risk of the plants they trade *(established but incomplete)* {5.6.2.1}, in some cases the business sector has developed voluntary codes of conduct in tandem with government regulations **(Box SPM.1)** *(well established)* {5.4.1, 6.3.1.4(4), Box 6.7}. It should be noted, however, that voluntary codes of conduct are intended to complement, not replace, obligations within national legislation that regulate activities that transport, sell or use alien species *(well established)* {6.3.1.4(4)}. The transport of invasive alien species along trade supply chains (e.g., in shipping containers) may be poorly managed and consequently may constitute a biosecurity risk *(well established)* {5.6.2.2}. There are many reasons for the limited adoption, implementation and efficacy of policy instruments, including varying capacity and resources across regions *(well established)* {6.2.2(7), 5.6.2.2} and lack of coordination, with unclear roles and responsibilities among government agencies, stakeholders and Indigenous Peoples and local communities *(well established)* {6.2.2(3), 6.2.2(7), 6.2.3, 6.7.2.5}. Nearly half of all countries (45 per cent) do not invest in management of biological invasions (Sustainable Development Goal indicator 15.8.1) *(established but incomplete)* {6.1.3}. Lack of awareness of the need for collective and coordinated responses can also hinder implementation {6.1.1, 6.2.2(9)}.

B. Globally, invasive alien species and their impacts are increasing rapidly and are predicted to continue rising in the future

B9 Intentionally or not, many human activities facilitate biological invasions globally *(well established)* {3.1.1, 3.2, 3.3, 3.4}. The transport and introduction of an invasive alien species can be intentional or unintentional, or in some cases both *(well established)* {3.2, 3.3}. Historically, many invasive alien species have been intentionally introduced outside their natural range around the world for their perceived benefits to people, without consideration or knowledge of their negative impacts *(well established)* {3.2.1, 3.2.3, 3.3.1, 3.3.2}. For example, invasive alien species are often used in forestry, agriculture, horticulture, aquaculture and as pets *(well established)* {3.2.3.2, 3.3.1.1}.[9] In the Mediterranean basin alone, more than 35 per cent of alien freshwater fish have arisen from aquaculture *(well established)* {3.3.1.1.1}. Invasive alien species have also been intentionally introduced for recreation and amenity *(well established)* {3.2.1, 3.2.3.3} and for soil stabilization *(well established)* {3.3.1.1.2, 3.3.1.6, 3.3.4.6}. Many invasive alien species have also been introduced unintentionally, including as contaminants of soils and traded goods, stowaways in shipments *(well established)* {3.2.3.1, 3.2.3.2, 3.2.3.4}, stowaways in ballast water and sediments, and as biofouling organisms that attach themselves to ships' hulls and other surfaces on vessels *(well established)* {3.2.3.1, 3.2.5, 3.3.2.3, Box 3.7}. Additionally, online trade in animals, plants and other organisms is contributing to the introduction of invasive alien species worldwide *(well established)* {2.1.2, 3.2.4.2}. Progressive degradation of nature, including from pollution and fragmentation of ecosystems, has facilitated the establishment and spread of invasive alien species *(well established)* {3.3.1.2, 3.3.1.3, 3.3.1.5, 3.3.1.6, 3.3.3}. Demographic drivers[10] also facilitate the introduction and spread of invasive alien species, although it is acknowledged that drivers differ across regions *(well established)* {3.2.2}. In the last 50 years, the number of people in the world has more than doubled, consumption has tripled, and global trade has grown nearly tenfold, with shifting patterns across regions *(well established)* {3.1.1}. This acceleration of the world economy is increasing the rate and magnitude of many direct and indirect drivers, particularly those related to trade, travel and land- and sea-use change,[11] leading to further biological invasions *(well established)* {3.1.1, 3.2.2}.

B10 The number of alien species is rising globally at unprecedented and increasing rates (Figure SPM.4) *(well established)* {2.2.1}. Thirty-seven per cent of all known alien species have been reported since 1970

9. IUCN. 2017. *Guidance for interpretation of CBD categories on introduction pathways. Technical note prepared by IUCN for the European Commission.* Available at: https://www.cbd.int/doc/c/9d85/3bc5/d640f059d03acd717602cd76/sbstta-22-inf-09-en.pdf

10. Demographic drivers have been identified by the IPBES Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services as one of the indirect drivers of change in nature, as described in Table 3.1

11. IPBES (2022). *The Thematic Assessment Report on the Sustainable Use of Wild Species of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Fromentin, J.M., Emery, M.R., Donaldson, J., Danner, M.C., Hallosserie, A., Kieling, D., Balachander, G., Barron, E.S., Chaudhary, R.P., Gasalla, M., Halmy, M., Hicks, C., Park, M.S., Parlee, B., Rice, J., Ticktin, T., and Tittensor, D. (eds.). IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.6425599

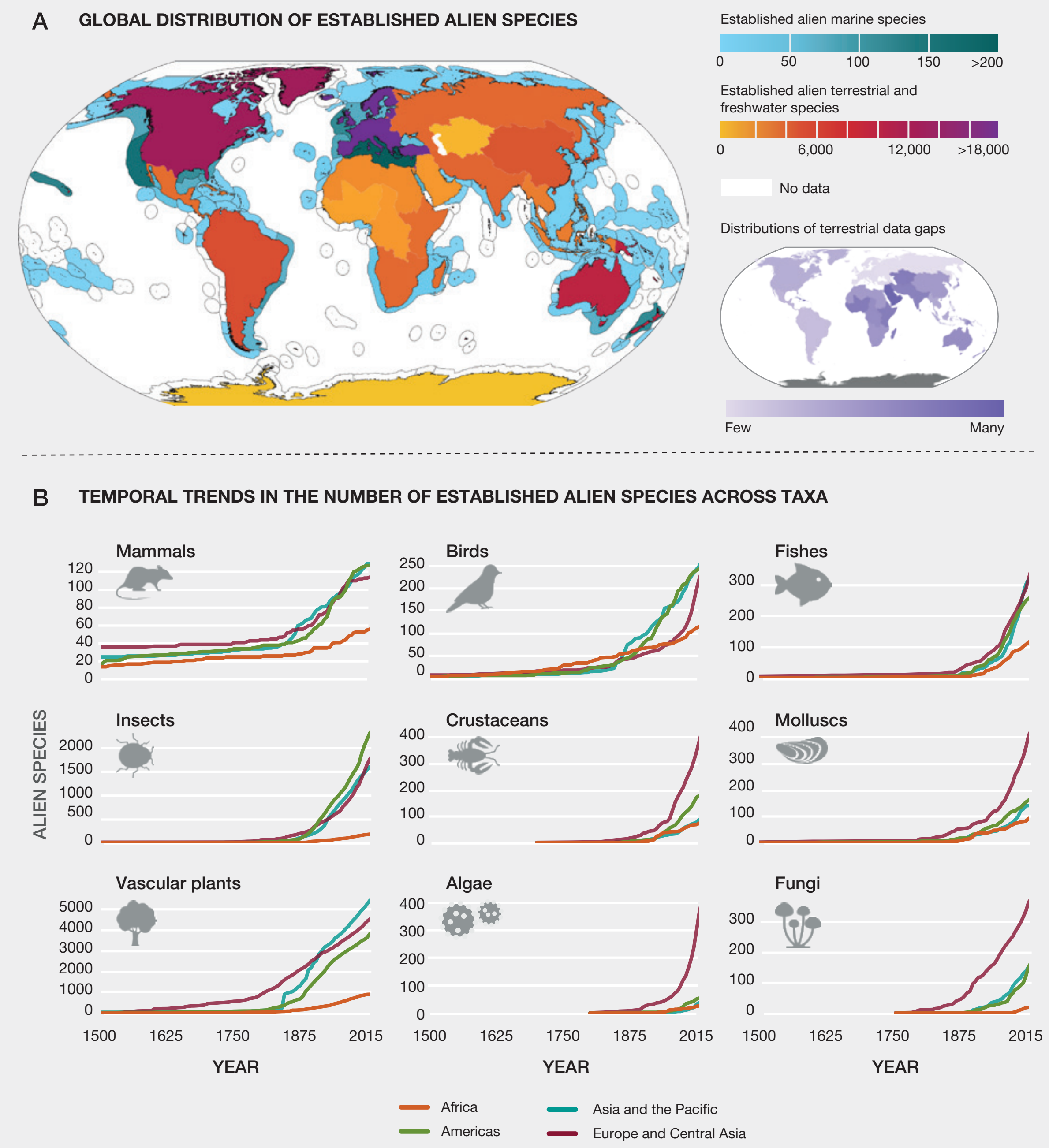

Figure SPM 4 **Global distribution and temporal trends in established alien species.**

(A) Total numbers of established alien species (terrestrial and freshwater) in the 18 IPBES subregions and marine ecoregions (marine) are indicated. White denotes missing information {2.2}. A gap analysis was conducted to identify data gaps for terrestrial regions, which are indicated in the inset {2.1.4, 2.2.3}. The data gap analysis could not be done for marine regions (white) and Antarctica (grey). **(B)** The temporal trends in the number of established alien species from 1500 to 2015 are shown for mammals, birds, fishes, insects, crustaceans, molluscs, vascular plants, algae and fungi, for the four IPBES regions {2.1.4, 2.4.1}.

(Figure SPM.3) *(established but incomplete)* {2.2.1}. The number of alien species has been rising continuously for centuries in all regions *(well established)* {2.2.1} and is expected to continue increasing in the future *(well established)* {2.6.1}. Global exploration and colonialism beginning in 1500, with the associated movement of people and goods, and industrialization from 1850 resulted in the transport and introduction of alien species and were historically important. Increases in global trade since 1950 have resulted in unprecedentedly high and increasing numbers of alien species being introduced **(Figure SPM.4)**. Some of these have become invasive *(well established)* {2.1, 3.2.3}. Even without the introduction of new species, given the opportunity, many already-established alien species in a region may continue to expand their geographic ranges and spread into new countries and regions *(well established)* {2.6.1}, including into remote environments such as mountain, polar (i.e., Antarctica and the Arctic) and desert ecosystems *(well established)* {2.5.2.8, 2.5.2.7, Box 2.7, Box 3.11}. Under a "business-as-usual" scenario, which assumes the continuation of past trends in drivers, the total number of alien species is projected to further increase globally, and by 2050 is expected to be approximately 36 per cent higher than in 2005 *(established but incomplete)* {2.6.1}. As trends in major drivers are predicted to accelerate in the future *(well established)* {3.1.1}, the number of alien species worldwide is expected to increase faster than predicted under the "business-as-usual" scenario *(established but incomplete)* {2.6.1}. There is a lack of quantified projections for invasive alien species under different scenarios **(Table SPM.A1)**, which impedes a comparison of trends for alternative futures *(well established)* {2.6.5}. Projections of long-term trends for invasive alien species numbers are not available but they are expected to be similar to those for established alien species *(established but incomplete)* {2.2.1}. The documented global economic cost of biological invasions has increased fourfold every 10 years since 1970 **(Figure SPM.3)** and is anticipated to continue rising *(established but incomplete)* {Box 4.13}.

B11 The increase in the transport and introduction of invasive alien species worldwide is primarily influenced by economic drivers, especially the expansion of global trade and human travel (Figure SPM.5) *(well established)* {2.1.2, 3.1.1, 3.2.3}. There has been a fivefold increase in the size of the global economy over the last 50 years *(well established)* {3.1.1}. International trade, which has increased nearly tenfold over the same period, represents the most important pathway through which invasive alien species are transported worldwide **(Figure SPM.5)** *(well established)* {3.1.1, 3.2.3.1}. There is a strong link between the volume of commodity imports and the number of invasive alien species in a region, and patterns in the global spread of species mirror shipping and air traffic networks *(well established)* {3.2.3.1}. The construction of shipping canals (e.g., Suez, Panama) has connected previously separated marine and freshwater regions, facilitating the spread of invasive alien species through species migration, ballast water transfers **(Box SPM.2)** and biofouling *(well established)* {3.2.3.1, 3.3.1.3}. For example, 150 years after the opening of the Suez Canal, marine alien species are still being newly recorded in the Mediterranean Sea *(well established)* {Box 3.7}. Biosecurity measures at international borders have not kept pace with the growing volume, diversity and origins of global trade (including e-trade) and travel *(well established)* {3.2.4.2, 3.2.3.4, 5.6.2.2}. Projected growth in international trade and the movement of people, including tourism, will lead to further pressure on border inspection regimes and could soon overwhelm the biosecurity capability of most countries *(well established)* {3.2.3.1, 6.3.1.4}.

B12 Accelerated establishment and spread of invasive alien species within countries are primarily driven by direct drivers, notably changes in land- and sea-use (Figure SPM.5) *(well established)* {2.2.1, 3.3.1, 3.6.2}. Land- and sea-use changes may increase the vulnerability of natural ecosystems to the establishment and spread of invasive alien species through increasing fragmentation and habitat disturbance, for example by changing grazing, fire regimes, soil disturbance, or watershed flow *(well established)* {3.3.1.2, 3.3.1.5}. Transportation and utility infrastructures such as roads, tracks, railways, pipelines, canals and bridges, among others, can create corridors that facilitate the spread of invasive alien species, including into remote, undisturbed and protected areas *(well established)* {3.3.1.3, Box 2.7, Box 3.7}. Marine and aquatic infrastructure may alter seascapes and the functioning of marine ecosystems, facilitating the spread of invasive alien species *(established but incomplete)* {3.2.2.4, 3.3.1.4, 5.6.1.4}. The numbers of invasive alien species were reported to be 1.5 to 2.5 times higher on pontoons and pilings than on natural rocky reefs *(established but incomplete)* {3.3.1.4}. More generally, land-use change can facilitate biological invasions through alteration of processes that cause natural disturbance of landscapes, such as wildfire or grazing regimes *(established but incomplete)* {3.3.1.5}. In several regions of the world, grazing by feral alien ungulates (horses, camels, buffalo, pigs) facilitates the spread of invasive alien plants, sometimes through complex species interactions involving the suppression of native species and the facilitation of other alien species *(well established)* {3.3.1.5.1}. As a specific example, invasive alien ungulates (wild boar, deer) can transport invasive ectomycorrhizal (root associated symbiotic) fungi, which are beneficial for the establishment and spread of alien pine trees, over long distances, rendering habitats susceptible to pine invasion *(well established)* {Box 3.10}. Climate change, along with the continued intensification and expansion of land-use change may lead to future increases in the establishment and spread of invasive alien species in disturbed habitats and in nearby natural habitats *(established but incomplete)* {3.3.4}.

B13 No driver acts in isolation, and interactions among drivers are amplifying biological invasions, leading to outcomes that can be difficult to predict *(well established)* {2.6.1, 3.1.5, 3.5}. The outcomes of interactions among multiple drivers, including feedback, are complex and varied *(well established)* {1.3.3, 3.1.5, 3.5}. Some of the highest current rates and greatest magnitudes

Box SPM 2 **The International Convention for the Control and Management of Ships' Ballast Water and Sediments: an example of international collaboration to prevent biological invasions.**

Many invasive alien species have been introduced to coastal and inland water ecosystems globally through ballast water discharges {3.2.3.1}. For example, following its introduction its introduction *via* ballast water discharge, *Dreissena polymorpha* (zebra mussel) has become widespread in the Great Lakes of North America {Box 2.9}. *Dreissena polymorpha* has been implicated in the transfer of botulinum toxin to higher trophic levels, which has been further facilitated by climate change, specifically by increased water temperatures, leading to mortality of waterfowl in the Great Lakes {Box 4.5}. Furthermore, the shells of *Dreissena polymorpha* can cause skin injuries to recreational swimmers and commercial fishers {Box 4.15}. The International Maritime Organization has developed an international instrument to address the transfer of harmful aquatic organisms and pathogens in ballast water of maritime vessels {5.5.1}. The International Convention for the Control and Management of Ships' Ballast Water and Sediments was adopted by the International Maritime Organization in 2004 and came into force in 2017 {5.5.1}. It is the first international legally binding legislation requiring ships to manage their ballast water so that aquatic organisms and pathogens are eliminated before the ballast water is released in a new location {3.2.3.1, 5.5.1, 6.1.3, 6.31}. While the global efficacy of ballast water management cannot be assessed yet, there is evidence that it has reduced invasive alien species introductions in the Great Lakes of North America {5.5.1}: between 1959 and 2006, one new alien species was discovered every seven months, but there was an abrupt shift (85 per cent decline) in the number of newly established alien species following the implementation of the ballast water regulations by Canada and the United States of America in 2006 and 2008 respectively {Box 2.9}.

Dreissena polymorpha (zebra mussel) was introduced through ballast water discharge in the Great Lakes of North America, causing a negative impact on nature, nature's contributions to people, and good quality of life.
Photo credit:Thirdwavephoto, WM Commons - CC BY 4.0

of biological invasion occur where land-use change interacts with one or more additional drivers *(established but incomplete)* {3.5.1, 3.5.2, 3.5.3}. For example, interactions among land-use change, climate change and nutrient pollution have driven the introduction, establishment and spread of *Pontederia crassipes* (water hyacinth) across Africa *(well established)* {Box 3.12}. Extraction of natural resources is closely linked with major economic and demographic drivers and can lead to a range of wider ecosystem impacts, including habitat degradation and loss, which facilitates invasive alien species *(well established)* {3.3.2, 3.4.2}. Climate change is predicted to lead to major changes in land- and sea-use and, in some regions, in human migration patterns *(established but incomplete)* {3.3.4}, but also to more extreme events among natural drivers, such as droughts, floods, wildfires, tropical storms and oceanic storm waves *(established but incomplete)* {3.3.4.3}. Additionally, invasive alien plants, especially trees and grasses, can sometimes be highly flammable and therefore promote more intense and frequent fire regimes, causing increased risks to nature and people and increased carbon release into the atmosphere *(well established)* {Box 1.4}. Climate change is also predicted to enhance the competitive ability of some invasive alien species and to extend areas suitable for them thus offering new opportunities for introductions and establishment *(established but incomplete)* {3.3.4}. Invasive alien species can facilitate the establishment and spread of other invasive alien species, resulting in positive feedback that increases impacts through a process known as "invasional meltdown" *(well established)* {3.3.5.1}. Biodiversity loss can reduce the resilience of ecosystems to invasive alien species, with

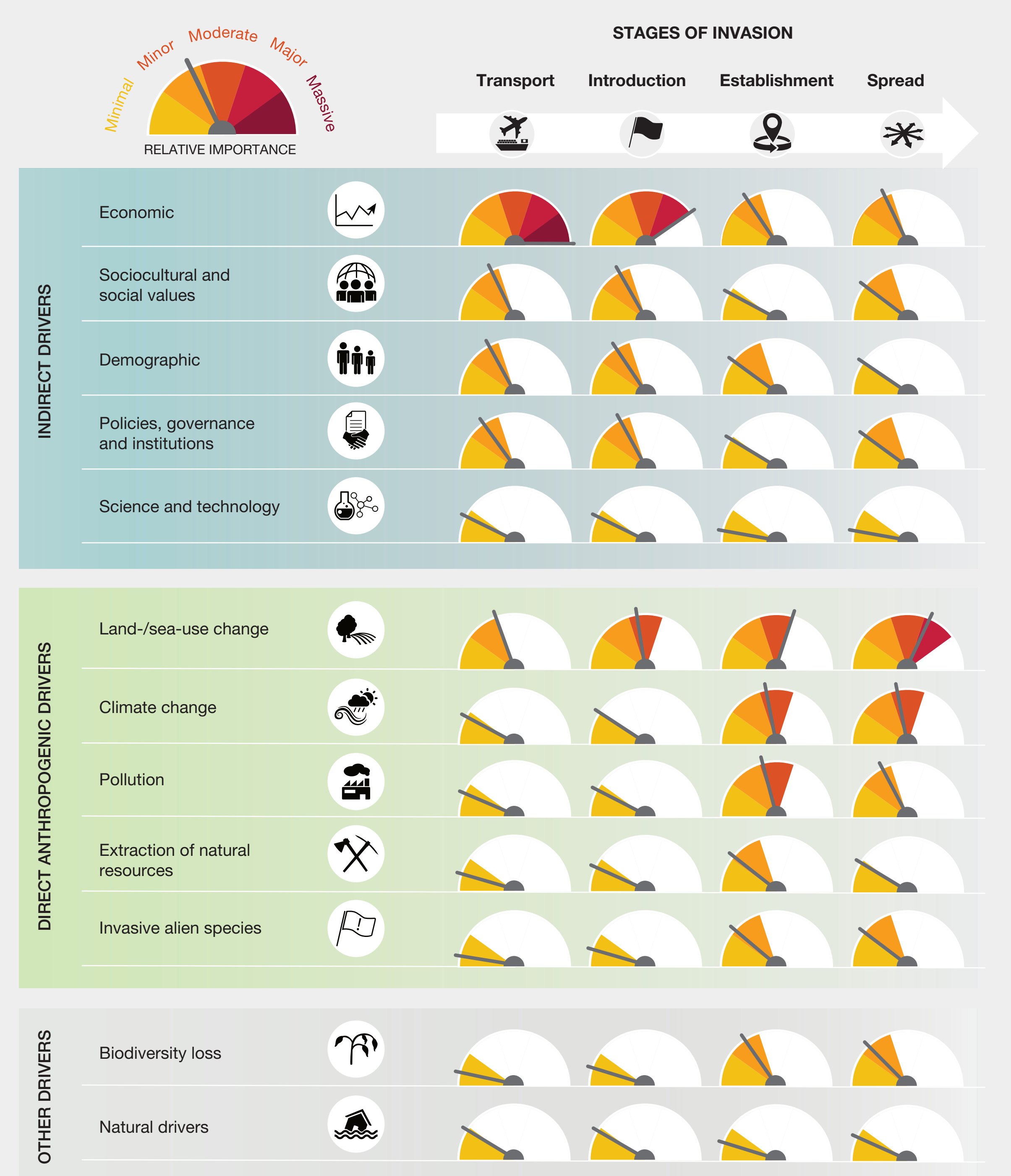

Figure SPM 5 **Relative importance of different drivers of change in nature in facilitating biological invasions across biomes per different stages of the biological invasion process (transport, introduction, establishment and spread), as determined through expert assessment, based on the evidence in chapter 3 {3.6.2}.**

These estimates are summarized across ecosystems and terrestrial biomes at the global scale. Drivers are classified according to the IPBES conceptual framework as direct or indirect drivers {3.1.3, Table 3.1}. Additionally, other drivers are included, namely biodiversity loss and natural drivers, as they can increase native ecosystem vulnerability or in other ways facilitate biological invasions {3.1.3}. Note that the role of invasive alien species as a driver refers to their role in facilitating other invasive alien species {3.3.5} and that this analysis focuses on the unintended consequences of policies, governance, institutions and technologies in facilitating biological invasions {3.2.4, 3.2.5}. The relative importance of drivers for each stage of the biological invasion process accounts for multiple, interacting, and non-additive effects of drivers, with differences in the overall importance of drivers across stages. While all drivers can potentially influence each biological invasion stage, indirect drivers, particularly those associated with economic growth, are more important in facilitating the transport and introduction stages {3.6.2}. In contrast, direct drivers, particularly land- and sea-use and climate change, are proportionally more important in facilitating the later stages of biological invasion {3.6.2}.

subsequent feedback facilitating the establishment and spread of other invasive alien species *(well established)* {3.4.2}. Indirect drivers also interact with one another. For example, sociocultural changes may lead to increased rates of infrastructure development through urbanization, and these interactions may further influence the rate and magnitude of change in land- and sea-use and other direct drivers that may in turn facilitate biological invasions *(well established)* {3.2.1}. Feedback and non-linear relationships among interacting drivers are likely to be exacerbated with ongoing and concurrent amplification of drivers *(established but incomplete)* {3.1.1, 3.5, 3.6.3, Box 4.5}, potentially leading to numbers of invasive alien species never previously encountered *(established but incomplete)* {2.6.1}.

B14 **Negative impacts of invasive alien species can occur long after first introduction, and currently observed threats from invasive alien species can lead to an underestimation of the magnitude of the future impact *(well established)* {1.4.4, 2.2.1}.** There are often time lags in detection and reporting of newly introduced invasive alien species *(well established)* {2.2.1}. Some invasive alien species spread very rapidly, while others take longer to spread and fully occupy their potential ranges *(well established)* {2.2.1, 2.2.3}. For some invasive alien species, the impact is immediate and continues into the long-term (e.g., fast-spreading pathogens such as Zika virus and *Batrachochytrium dendrobatidis* (chytrid fungus), and fast-spreading predators such as lionfish), while for others there may be a considerable time lag, spanning decades in some cases, before the impact is apparent (e.g., many invasive alien trees) *(well established)* {1.5}. Such time lags can lead to people not perceiving the ongoing slow changes in their environment, including the impacts of invasive alien species *(well established)* {1.5.2}. There can also be significant time lags in the response of invasive alien species to various drivers because the underlying processes that facilitate biological invasions operate at varying temporal scales (short- to long-term) *(well established)* {1.5, 3.2.3.1, 3.6.3}. Invasive alien species may increase in numbers after a long period at low density as a result of changes in interactions with other species, for example as a result of the introduction of a missing dispersal agent or the removal of a competitor (3.3.5.1). For example, in the western United States, the invasive alien *Carcinus maenas* (European shore crab) reduced the abundance of a native clam, releasing another alien species, *Gemma gemma* (the amethyst gem clam), from competition, allowing it to become superabundant and to spread, after having been locally distributed and at low abundance for over 50 years *(well established)* {3.3.5.1}. Patterns in the numbers of alien species seen today reflect the drivers of decades ago (i.e., invasion debt) *(established but incomplete)* {3.1.1, 3.1.5}. Consequently, past and ongoing amplification of drivers may lead to a long legacy of future invasive alien species as, for example, the number of new alien species that become invasive increases over time (i.e., invasion debt) *(established but incomplete)* {2.3.1.5, 3.1.5, 3.6.3}.

C. Invasive alien species and their negative impacts can be prevented and mitigated through effective management

C15 **Management of invasive alien species has been successful in many contexts (Figure SPM.6, Table SPM.1) *(well established)* {5.5.1, 5.5.2, 5.5.3, 5.5.4, 5.5.5, 5.5.6}.** There are three options for preventing or reducing the number and negative impacts of invasive alien species:

- Pathway management, based on the analysis of pre-border, border and post-border risks, can prevent the movement and spread of invasive alien species through surveillance and the implementation of biosecurity response measures *(well established)* {5.3.1.1, 5.5.1, 5.5.2}.
- Species-based management at a local or landscape level, which includes surveillance, early detection and rapid response, eradication, containment and widespread control (including biological control), and can be applied throughout the biological invasion process *(well established)* {5.3.1.2, 5.5.2, 5.5.3, 5.5.4, 5.5.5}.
- Site- or ecosystem-based management, which can both protect and restore native species and ecosystems *(well established)* {5.3.1.3, 5.5.6}.

The use of individual species-based and site-based approaches for the management of multiple invasive alien species has been both successful and cost-effective for terrestrial and closed water systems, especially in biogeographically isolated areas such as small islands and lakes *(well established)* {5.3.1, 5.3.2, 5.5.4}. While some management approaches can be applied at multiple scales across terrestrial and closed water systems *(well established)* {5.1.1, 5.3.1.4.}, pathway management (e.g., ballast water and biofouling; **Box SPM.2**) is by far the

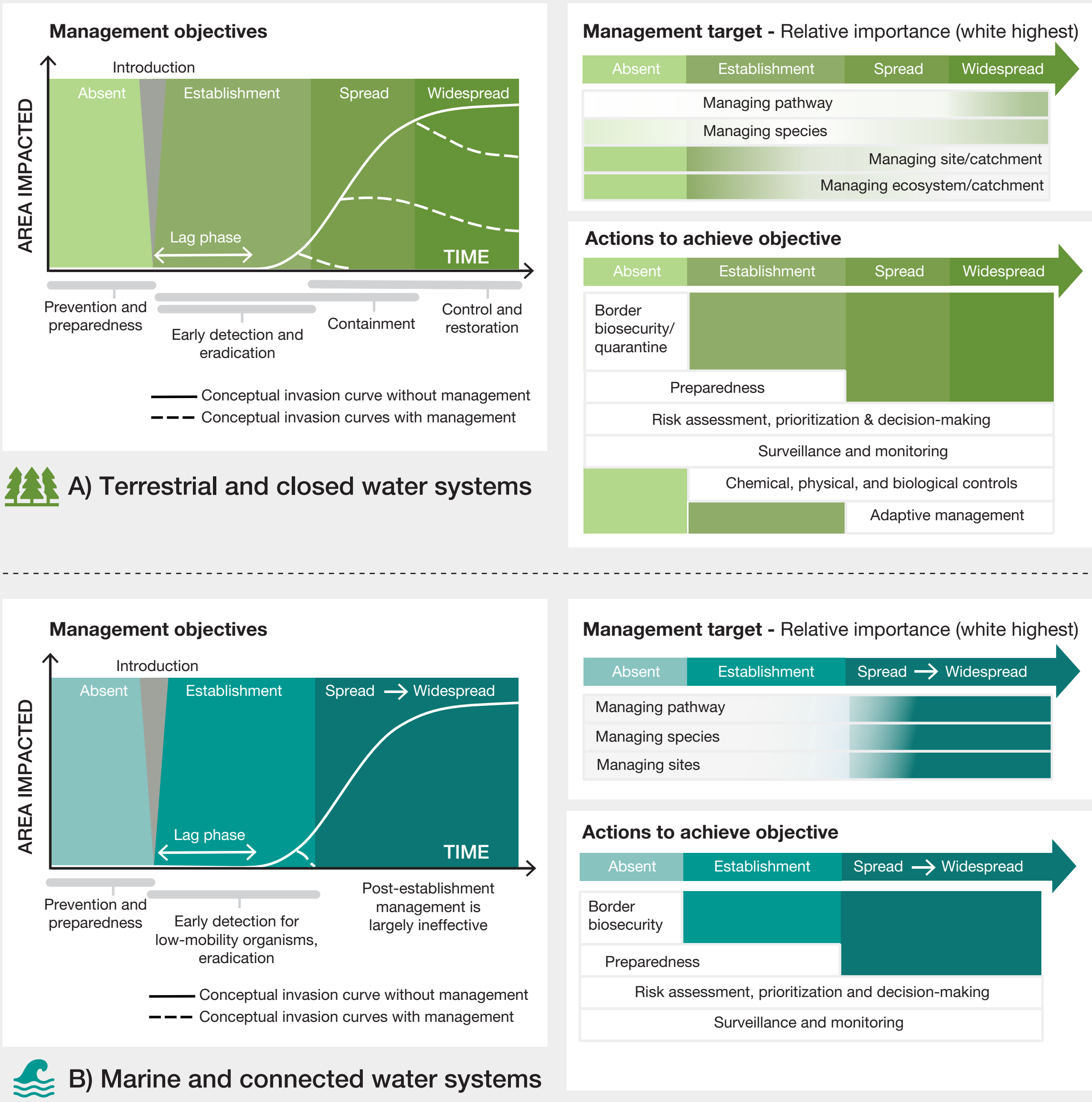

Figure SPM 6 **Conceptual diagram of management–invasion continuum.**

Management objectives panels A and B show the generalized invasion curve without management and the expected changes in the trajectory of the invasion curve with appropriate management actions in (A) terrestrial and closed water systems (including lakes and coastal systems such as salt marshes) and (B) marine and connected water systems (including rivers). Post-establishment management actions (such as containment and control) are not shown in panel B as they are generally not achievable in such systems. In a management context, the first detection (introduction point), lag phase and exponential spread phase are important points at which to implement an early detection and rapid response management plan. This figure is conceptual, and the curves do not represent actual population dynamics of invasive alien species. In the **Management target** panels, the white boxes indicate the optimal management options at each stage of the biological invasion process. The colour gradient of the managing pathway, managing species, managing site and managing ecosystem boxes show how the relative importance of each changes as a biological invasion progresses (managing ecosystems is not applicable in marine and connected water systems). In the **Actions to achieve objective** panels, the white boxes indicate the typical management actions needed to achieve each management objective.

Table SPM 1 **Objectives and actions for managing biological invasions.**

Objectives and actions for managing biological invasions within terrestrial and closed water systems or marine and connected water systems and the level (high, medium, low) of their (a) current availability (availability of target-specific tools for implementing management); (b) ease of use (ease of implementation or specialist or technological expertise to implement); and (c) effectiveness (likely long-term efficacy and outcomes of implementation). Hashed boxes indicate responses with a low level of confidence and crossed boxes indicate there was no data available to perform an assessment. Actions are aligned with **Figure SPM.6** and encompass pathway management, species-, site- and ecosystem-based management targets. All management approaches may have non-target impacts, as indicated by superscript a.

OBJECTIVES	MANAGEMENT ACTIONS	TERRESTRIAL AND CLOSED WATER SYSTEMS			MARINE AND CONNECTED WATER SYSTEMS		
		Current availability	Ease of use	Effectiveness	Current availability	Ease of use	Effectiveness
Prevention and preparedness	Horizon scanning	High	Medium	High	Medium	Medium	Low
	Import controls and border biosecurity	High	Medium	Medium	Medium	Low	Low
	Pathway management	Medium	Low	Low	Medium	Low	Medium
	Risk analysis	High	High	High	High	Low	Medium
Early detection	Surveillance	Medium	Low	High	Low	Low	High
	Diagnostics	High	Medium	High	Low (hashed)	Low (hashed)	Medium (hashed)
Eradication	Physical eradication[a]	High	Low	Low	Low	Low	Low
	Chemical eradication[a]	High	Medium	Medium	Low	Low	Low
	Adaptive management	Medium	Low	High	Low (hashed)	Low (hashed)	Low (hashed)
Containment and control	Physical control[a]	High	Low	Low	Low	Low	Low
	Chemical control[a]	High	Medium	Medium	Low	Low	Low
	Biological control[a]	High	Medium	High	No data (crossed)	No data (crossed)	No data (crossed)
	Adaptive management	Medium	Low	High	Low	Low	Low
Ecosystem restoration	Adaptive management	High	Low	High	Low	Low	Low
Public understanding	Public engagement	High	Medium	High	High	Low	High

Hashed boxes indicate a low level of confidence in the assessment

Crossed boxes indicate no data was available to perform an assessment

Column values

High Medium Low

most effective option for managing biological invasions in marine and connected water systems, and can be achieved by enhanced international and regional cooperation *(well established)* {5.5.1, 6.3.2.2}.

C16 There are effective decision-making frameworks and tools that can support management of biological invasions (Table SPM.1) *(well established)* {5.2.1, 5.2.2}. Frameworks and tools have been developed based on evidence from practice, science and other knowledge systems, including those of Indigenous Peoples and local communities. These can underpin impact assessment, monitoring and prioritization of intentional and unintentional introduction pathways, species and sites for the successful management of biological invasions *(well established)* {5.2.2}. Although many knowledge and data gaps exist **(Table SPM.A1)**, the tools enable management actions to proceed under a risk assessment and risk management framework in line with a precautionary approach, as appropriate, using inclusive decision-making that leads to the review of all the measures *(well established)* {5.2.2.1, 5.2.2.3, 5.2.2.4, 5.3.3, 6.4.1}. Decision-making may be challenged by multiple sources of uncertainty, such as projections in other drivers of change, which can be recognized, quantified and documented to contextualize decisions *(well established)* {5.6.2.5}. Many sources of accessible literature and information (including open-access data), analytical tools and other types of knowledge can be used to support decision-making for all countries, which could lead to coordinated management outcomes globally **(Table SPM.A3)** *(established but incomplete)* {6.6.1.5}.

C17 Preventing the introduction of invasive alien species is the most cost-effective management option (Figure SPM.6) *(well established)* {5.5.1}. Prevention measures through pathway management, including strictly enforced pre-border quarantine, import controls and border biosecurity, have increased interception rates and slowed the rate of invasive alien species arriving and establishing globally *(well established)* {5.4.3.1, 5.5.1}. For example, in Australasia, the number of interceptions of *Halyomorpha halys* (brown marmorated stink bug), recognized as a major threat in the agricultural sector, have declined following implementation of a systems-based pathway management approach *(well established)* {5.5.1}. Measures to address escape from confinement are also necessary *(established but incomplete)* {5.3.1.1}. It is, however, difficult to prevent further natural dispersal of invasive alien species from a previously invaded range *(well established)* {5.5.1, Box 1.6}. Prevention is important on islands and in ecosystems where eradication poses significant technical challenges *(well established)* {5.3.2}. Effective prevention measures depend on adequate and sustained funding, capacity-building, technical and scientific cooperation, transfer of technology, monitoring, and relevant and appropriate biosecurity legislation and enforcement, which is supported by strong infrastructure, quarantine and inspection facilities, including diagnostic support services *(well established)* {5.4.2, 5.6.2, 5.6.2.2, 5.7}. Risk assessment could be used by businesses to engage different sectors in the prevention and management of biological invasions *(established but incomplete)* {5.6.2.1}. Adoption of regulated species lists with explicit prohibition of or permission for the importing of specific alien species, underpinned by risk analysis, has been an effective prevention strategy *(well established)* {5.6.2.1, 6.3.1.4}. It is estimated that nearly 70 per cent of marine invasive alien species established worldwide were introduced *via* biofouling *(established but incomplete)* {5.5.1}.

C18 When prevention fails or is not possible, preparedness, early detection and rapid response are effective at reducing rates of invasive alien species establishment in terrestrial and closed water systems, and critical for marine and connected water systems *(well established)* {5.4.2, 5.5.1, 5.5.3, 5.5.2, 5.6.3.3}. Horizon scanning and risk analysis are examples of the many decision-support tools used to identify and prioritize emerging invasive alien species to support preparedness *(well established)* {5.2}. Such tools can inform the development of rapid response plans in advance of an incursion to guide action effectively following the detection of priority invasive alien species *(well established)* {5.2.2.1.a, 5.2.2.1.b, 5.5.1}. Early detection of invasive alien species can enable rapid intervention to contain and eradicate invasive alien species before they spread *(well established)* {5.1.1, 5.3.1.1, 5.5.2}. General surveillance strategies (e.g., through citizen science, sentinel sites, and remote sensing) for detecting new invasive alien species can also underpin effective preparedness *(established but incomplete)* {5.3.1.1, 5.4.2.1.a, 5.4.2.2.a, 5.5.2, Box 6.20}. For example, in Africa, Asia and Latin America, the PlantwisePlus programme assists smallholder farmers with the identification of pests and damaged crops, contributing to early detection of invasive alien species outbreaks *(well established)* {5.5.2}.

C19 Eradication has been successful and cost-effective for some invasive alien species, especially when their populations are small and slow-spreading in isolated ecosystems such as islands *(established but incomplete)* {5.5.3}. Over the last 100 years, there have been 1,550 documented examples of eradication on 998 islands, with success cited in 88 per cent of cases *(well established)* {5.5.3}. One of the many examples is French Polynesia, where *Rattus rattus* (black rat), *Felis catus* (cat), *Oryctolagus cuniculus* (rabbit) and *Capra hircus* (goat) have been successfully eradicated *(well established)* {Box 5.8}. Eradication of invasive alien plants is particularly difficult because of the longevity of dormant seeds that can persist in soil (i.e., soil seed bank), although there are examples of successful eradication of invasive alien plant species with limited distributions *(well established)* {5.5.3}. Also, rapid response to incursions, detected early, of some invertebrates have been successful, for example, eradication of *Solenopsis invicta* (red imported fire ant) in New Zealand *(well established)* {Box 5.14}. There are examples of larger-scale eradications, such as *Ondatra zibethicus* (muskrat) and *Myocastor coypus* (coypu) from the United Kingdom *(well established)* {5.5.3}. However, large-scale eradications are difficult and unlikely to be feasible in many cases *(well established)* {5.5.3}. In addition to the extent of the area invaded, the success of eradication programmes depends on the support and engagement of relevant stakeholders and Indigenous Peoples and local communities *(well established)* {5.4.2.2.a, 5.5.3, 5.6.2.1, 5.6.2.2}. Eradication programmes are aided by a rapid flow of information on the extent and location of invasive alien species, which can be provided by people who live nearby *(well established)* {5.4.2.2.a, 5.5.3}. Evidence suggests that there have been no fully successful eradication programmes for established invasive alien species in marine ecosystems *(well established)* {5.5.3}. While eradication programmes can only be achieved with access to upfront cost, they are generally cheaper than long-term and permanent control cost and impacts *(well established)* {5.5.3}.

C20 When eradication is not possible for different reasons, invasive alien species can be contained and controlled, particularly in terrestrial and closed water systems *(well established)* {5.4.3, 5.4.4, 5.5.4, 5.5.5}. There are many examples of successful containment and control of invasive alien species in terrestrial and closed water systems and aquaculture (e.g., containment of *Styela clava* (Asian tunicate) invading the aqua-cultured blue mussel in Canada) *(well established)* {5.5.4}, but most attempts in marine and open water ecosystems have been largely ineffective *(established but incomplete)* {5.5.4, 5.5.5}. Containment of invasive alien species can be achieved with physical, chemical and biological control actions or in combination **(Table SPM.1)** *(well established)* {5.4.3.2, 5.5.4}. Physical and chemical control options are mostly effective at a local scale but can also be effective at larger scales; these control options are limited by labour costs and generally provide short-term suppression but not sustained control *(well established)* {5.4.3.2.a}. Furthermore, chemical control may have non-target impacts, needs to be implemented under regulatory compliance requirements and has decreasing societal acceptability *(well established)* {5.4.3.2.b}. Biological control has been very effective in controlling some invasive alien plants, invertebrates and, to a lesser extent, plant microbes and a few invasive alien vertebrates, but it may have non-target impacts if not well regulated *(well established)* {5.5.5.3}. To reduce the risks of unintended consequences, including non-target impacts, from biological control, international standards and risk-based regulatory frameworks (developed under the International Plant Protection Convention) have been applied and continue to be effective across many countries *(well established)* {5.5.2}. The use of biological control for invasive alien plants and invertebrates has been successful in more than 60 per cent of documented cases **(Box SPM.3)**, with one third of the alien plant species requiring no further form of control, while also leading to benefits to biodiversity and ecosystem resilience *(well established)* {5.5.5.3}. Classical biological control to suppress invasive alien species populations at landscape scales has been effectively practised for more than 100 years *(well established)* {5.5.5.3}.

C21 Adaptive management, including ecosystem restoration, can improve the management of invasive alien species and support the recovery of nature's contributions to people in terrestrial and closed water systems *(well established)* {5.3.3, 5.4.4.3a, 5.5.6, 5.7}. The integration of site- and/or ecosystem-based management, including ecosystem restoration, can improve management outcomes, enhancing ecosystem function and resilience to environmental change, including future invasive alien species, especially under climate and land-use change **(Box SPM.4)** *(well established)* {5.3.1, 5.3.2, 5.4.3, 5.5.6, 5.6.1.3}. The success of any applied adaptive site- or ecosystem-based

Box SPM 3

Classical biological control of *Mikania micrantha* (bitter vine): an example of effective suppression of a widespread invasive alien species.

Classical biological control uses host-specific natural enemies (biological control agents) of invasive alien species (target) to suppress and control such species. *Mikania micrantha* (bitter vine), a native species of Central and South America, is one of the highest-impact fast-growing {2.5.2.1} invasive alien plants within the agricultural systems and natural and planted forests of the Asia-Pacific region {Box 5.21}, affecting the livelihoods of farmers and rural communities, including women {4.5.1, 4.6.1}.

In the native range of *Mikania micrantha*, a rust fungus (*Puccinia spegazzinii*) specific to this invasive alien plant causes necrosis of leaves and cankers on the stem and petioles {Box 5.21}. Starting in 2006, *Puccinia spegazzinii* was introduced as a classical biological control agent and established in five countries in the Asia-Pacific region, where it has provided effective control of *Mikania micrantha* {Box 5.21}. However, in India the rust fungus failed to survive in the field following introduction {Box 5.21}.

Box SPM 4 **Working for Water programme: an example of management of invasive alien species leading to recovery of nature's contributions to people.**

Control of widespread invasive alien species requires sustained, large-scale efforts but can lead to improvement in the provision of a range of nature's contributions to people {Box 5.19}. Certain invasive alien plants, such as shrubs and trees, can reduce water availability, especially in scenarios of increasing drought caused by climate change {Box 5.4}. In South Africa, the Working for Water programme, an Expanded Public Works Programme, was introduced in 1995 and targeted historically disadvantaged communities, primarily women, youth and disabled people, creating jobs to reduce poverty nationally through the removal of widespread woody invasive alien species threatening water conservation {Box 5.19}. The programme generated 20,000 jobs per year over the first 15 years and has helped to improve nature's contributions to people by improving water security {Box 5.19}. It has contributed to rural development by providing training in entrepreneurial and management skills while encouraging a sense of community and dignity among workers, especially women. The Working for Water programme shows how partnerships with rural communities to manage invasive alien species can bring both ecological and social benefits {Box 5.19}.

management approach, including ecosystem restoration, depends on long-term monitoring to assess management efficacy using ecological and social indicators *(established but incomplete)* {5.5.2, 6.6.3}. Long-term monitoring of sites ensures early detection of new introductions, reintroductions and re-emergence of invasive alien species (e.g., from a seed bank that includes invasive alien plants) and can inform further management actions *(well established)* {5.4.3.3.b, 5.5.6}. However, most studies failed to quantify the effectiveness of ecosystem restoration since they failed to measure the initial status of native vegetation. This has led to inconsistent conclusions regarding the best invasive alien plant control option which may lead to the most effective ecosystem restoration {5.4.3.3b; 5.5.6}. Regarding freshwater ecosystems, monitoring biodiversity using macroinvertebrate-based indices is a widely used method globally. However, knowledge is lacking on how invasive alien species may affect the metric scores and therefore classification of a river's status *(established but incomplete)* {5.6.2.3}. In marine and connected water systems, ecosystem restoration has so far proved to be largely ineffective because the systems are open, leading to difficulties in implementing and evaluating management actions *(established but incomplete)* {5.5.6, 5.6.1.1}.

C22 **Tools and technologies increase efficiencies when managing biological invasions and controlling invasive alien species, with many new options emerging *(established but incomplete)* {5.4}.** The development of tools and technologies ranging from biotechnology to bioinformatics and data analytics is ongoing for managing pathways, surveillance and detection, rapid response and eradication, local containment and control of widespread invasive alien species *(well established)* {5.4.1, 5.4.2, 5.4.3}. eDNA-based approaches have been used for detection and identification of invasive alien, mostly aquatic, species such as *Orconectes rusticus* (rusty crayfish) *(well established)* {5.4.2.1}. New approaches can be integrated with existing management actions to support site- and ecosystem-based management and restoration *(established but incomplete)* {5.4}. Multi-stakeholder engagement, including risk communication and context-specific application of approaches through local communities, can improve public acceptability and adoption of new tools and technologies for managing biological invasions and the control of invasive alien species *(well established)* {5.2.1, 5.4.3, 5.6.2.1, 6.4.1}. Potential benefits and risks of novel technologies can be assessed using a risk assessment and risk management framework in line with a precautionary approach, as appropriate *(well established)* {5.4.3.2.f}. Using this framework in consultation with regulators, stakeholders and Indigenous Peoples and local communities can limit the potential for unintended consequences *(well established)* {5.4.3.2}. However, most countries do not have the regulatory frameworks and/or technical capabilities needed to guide and support development and implementation of new tools and technologies *(established but incomplete)* {5.4.3.2, 6.3.3.4}. Access to modern tools and technologies and the ability to utilize them can be limited, particularly in developing countries, meaning greater capacity-building is required and improved technical and scientific cooperation *(well established)* {5.6.2.4, 6.7.2.7}.

C23 **Stakeholder engagement, capacity-building and sustained resourcing are critical to the success of adaptive management *(well established)* {5.2.1, 5.6.2.1, 5.6.2.2, 5.6.2.4, 6.4.1, 6.5.3, 6.5.6, 6.5.7}.** Access to adequate and sustained financial and other resources, including international funding to support developing countries, underpins and improves the effectiveness of actions for long-term management of biological invasions, including eradication, control and ongoing monitoring, by, for example, providing access to modern tools and enhancing capacity to deploy them *(well established)* {5.3.1, 5.5.7, 5.6.2.1, 5.6.2.2, 5.6.2.4, 6.5, 6.5.7}. Engagement by all stakeholders, governments and the private sector helps to optimize management of biological invasions in terms of economic, environmental and social outcomes, particularly when resources are limited *(well established)* {5.2.1, 6.5.1}. Societal support is important for eradication and control of some invasive alien species, particularly vertebrates, for which there are ethical considerations {5.3.1.4, 5.4.3.2, 5.6.2.1}. A lack of stakeholder participation in adaptive management can lead to negative consequences for good quality of life, especially for Indigenous Peoples and local communities who have adapted by using invasive alien species, that include loss of livelihoods, marginalization and/or gender inequity *(well established)* {Box 4.18, 5.2.1, 5.4.3.3.a, 5.5.3, 5.6.1.2, 6.4.1}. The involvement of all stakeholders can be achieved by using an adaptive co-management approach to the process, from decision-making to the implementation of management actions *(well established)* {5.4.3.3.a, 5.6.2.5}. Adaptive co-management includes capacity-building; co-creation, co-design, co-development and co-implementation; social learning; and broad partnerships *(established but incomplete)* {5.7, 6.4.2, 6.4.3.2, 6.4.4}. Collaboratively addressing the management of biological invasions around which there are conflicting values among different sectors, stakeholders and Indigenous Peoples and local communities is a significant global policy challenge *(well established)* {5.6.1.2}.

C24 **The knowledge, practices, values and customary governance systems of Indigenous Peoples and local communities can improve management outcomes *(established but incomplete)* {5.2.1, 5.5.2, 5.5.4, 5.5.5, 5.6.1.2, 6.4.3}.** Many communities successfully manage invasive alien species on their lands *(established but incomplete)* {Box 5.6, 5.5.2, 5.5.4, 5.5.5}, leading to increases in nature's contributions to people **(Box SPM.4)** *(established but incomplete)* {5.5.4, 5.5.5}. Consultation with Indigenous Peoples and local communities, through their free, prior and informed consent, by applying co-design principles for decision-making and actions helps to ensure efficacy of management outcomes at the local level *(established but incomplete)* {5.2.1, 6.4.3}. Co-delivered biocultural management plans based on shared scientific, technical and Indigenous and local knowledge systems have assisted surveillance and detection, eradication, containment and control of invasive alien species *(established but incomplete)* {5.5.3, 5.6.1.2, 6.4.3.2}. Such co-governance structures improve quality of life for Indigenous Peoples and local communities *(established but incomplete)* {6.4.3}.

D. Ambitious progress to manage biological invasions can be achieved with integrated governance

D25 **Management of biological invasions and prevention and control of invasive alien species can be achieved through a context-specific integrated governance approach with a set of complementary strategic actions (Figure SPM.7) *(established but incomplete)* {6.2.3, 6.7.1, 6.7.2, 6.7.3}.** Integrated governance for biological invasions consists of establishing the relationships between the roles of actors, institutions and instruments. This involves all those elements of the interactions between people and nature that act on biological invasion and their management, in order to identify the strategic interventions needed to improve outcomes of prevention and control of invasive alien species {Box 6.5}. A context-specific integrated governance approach provides flexibility for countries to identify which strategic actions should be prioritized and can help in managing trade-offs and policy conflicts and in avoiding unintended policy consequences and inefficient expenditure *(established but incomplete)* {6.2.3, 6.7.1}. Strategic actions to prevent the introduction and impact of invasive alien species include:

1. Enhance coordination and collaboration across international and regional mechanisms *(established but incomplete)* {6.2.3.4, 6.7.2.1};
2. Develop and adopt effective and achievable national implementation strategies *(well established)* {6.2.3.2, 6.3.3.1, 6.7.2.3};
3. Share efforts and commitments and understanding of the specific roles of all actors *(established but incomplete)* {6.7.2.5};
4. Improve policy coherence *(well established)* {6.3.1.1, 6.3.2, 6.3.3.1, 6.7.2.2};
5. Engage broadly across governmental sectors, industry, the scientific community, Indigenous Peoples and local communities and the wider public *(established but incomplete)* {6.4.2, 6.4.3, 6.7.2.4};
6. Support, fund and mobilize resources for innovation, research and environmentally sound technology *(established but incomplete)* {6.3.3.4, 6.7.2.7};
7. Support information systems, infrastructures and data sharing *(established but incomplete)* {6.6.2.3, 6.7.2.6}.

Effective implementation, robustness of relevant institutions, responsiveness and equitability are key properties of

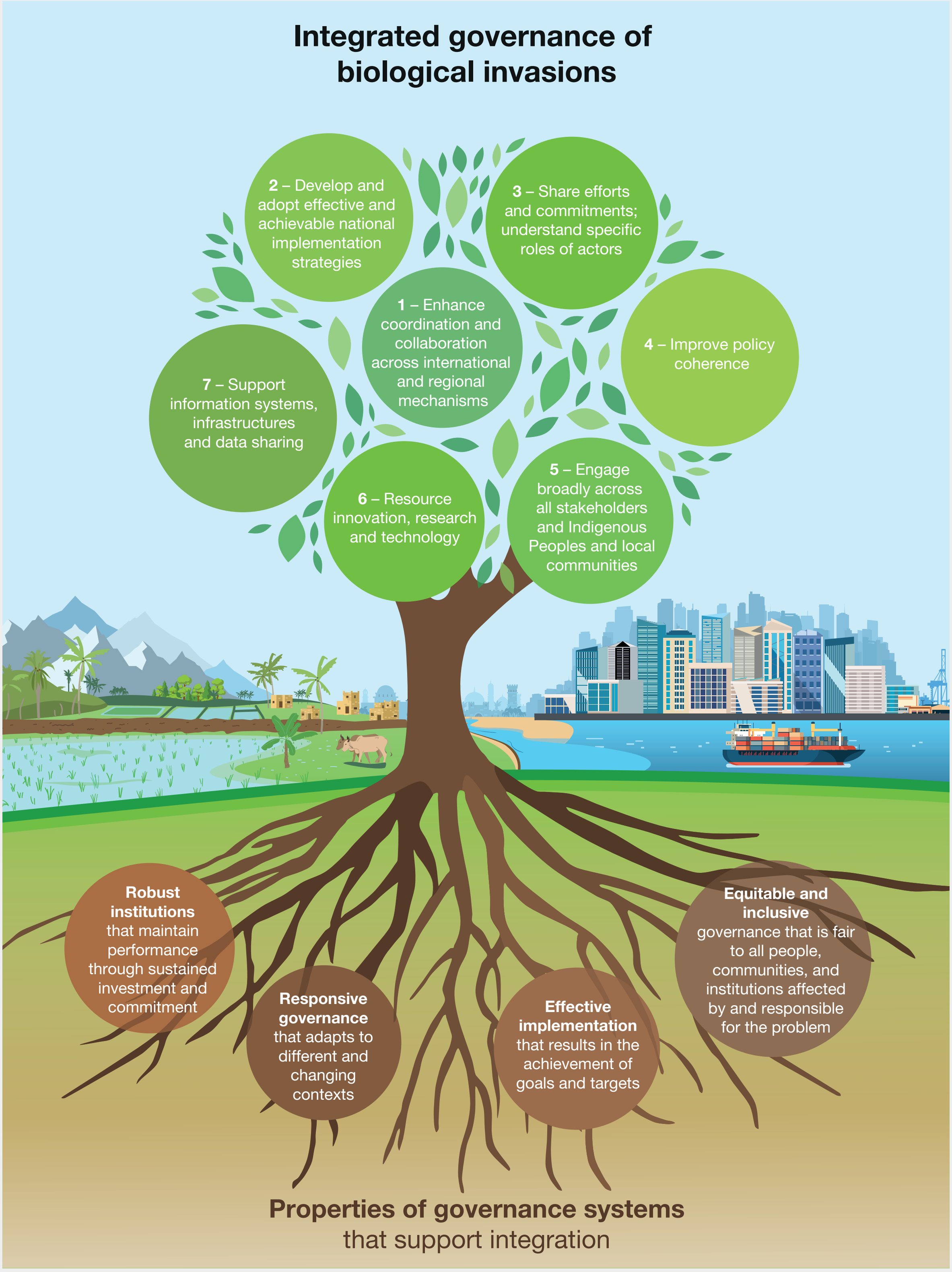

Figure SPM 7 **Integrated governance of biological invasions.**

A context-specific integrated governance approach to biological invasions is enabled by a governance system with properties that support integration, and a set of strategic actions that together are designed to bring about the progress needed to meet national and international goals and targets for biological invasions. Integrated governance is rooted in four essential properties of governance systems (tree roots) that support the strategic actions (branches) to be achieved. Together, the properties and actions will bring about the step change needed for effective and sustainable management of biological invasions. Integrated governance for biological invasions reinforces the enabling conditions identified as necessary to fulfil the 2030 mission of the Kunming-Montreal Global Biodiversity Framework. An integrated governance approach activates specific strategic actions that promote transformative change to meet the goals of preventing and controlling biological invasions.

The strategic actions are:

1. Enhance coordination and collaboration across international and regional mechanisms.
2. Develop and adopt effective and achievable national implementation strategies.
3. Share efforts, commitments and understanding of the specific roles of all actors.
4. Improve policy coherence.
5. Engage broadly across governmental sectors, industry, the scientific community, Indigenous Peoples and local communities and the wider public.
6. Support, fund and mobilize resources for innovation, research and environmentally sound technology.
7. Support information systems, infrastructures and data sharing.

The proposed strategic actions are enabled when the system-wide properties of governance (roots) are robust, equitable and inclusive, responsive and focused on effective implementation. The numbers on the branches do not imply a ranking.

governance systems that enable integrated governance **(Figure SPM.7)**, while the importance of context-appropriate solutions is acknowledged *(established but incomplete)* {6.2.3, 6.7.3}.

D26 One of the most effective ways to manage biological invasions is to develop coherent policy instruments that reinforce strategic actions across sectors and scales *(established but incomplete)* {6.3.1, 6.3.2, 6.5.4}. Many policy instruments aimed at preventing the introduction of invasive alien species have been adopted, including multilateral agreements, national laws, multi-level regulations and voluntary codes of conduct *(well established)* {6.1.2, 6.3.1}. They have jointly contributed to reducing the impacts of invasive alien species on nature, nature's contributions to people, and good quality of life *(established but incomplete)* {5.5.1, 6.1.3}. The work under various relevant international organizations, partnerships and multilateral environmental agreements (e.g., the Convention on Biological Diversity, the World Trade Organization, the International Maritime Organization, the International Plant Protection Convention, the World Organisation for Animal Health, the Convention on the Conservation of Migratory Species of Wild Animals and the Convention on International Trade in Endangered Species of Wild Fauna and Flora) is not adequately aligned to address the problem posed by invasive alien species *(well established)* {6.3.1.3, 6.3.1.4}. Enhanced coordination and collaboration across international and regional mechanisms are key strategic actions for rapid and transformative progress *(established but incomplete)* {6.7.2.1} and could help international, national and local agencies that implement policies for the environment, agriculture, aquaculture, fishing, forestry, horticulture, border control, tourism and trade (e.g., in wildlife, but also including online trade in other animals, plants and other organisms), community and regional development (including infrastructure), transportation and health deliver a coherent approach to biological invasions *(well established)* {6.3.1.1}. Such coordination and collaboration efforts would consider the trade-offs across sectors {6.3.1.1(2), 6.3.1.3}, stakeholders and Indigenous Peoples and local communities {1.5.1}, and the interdependence between invasive alien species and other drivers *(established but incomplete)* {3.1.1, 3.1.5, 6.2.3.2, 6.7.2.2}. Collaborative, multisectoral and transdisciplinary approaches (such as One Health) provide frameworks to prevent and control invasive alien species by strengthening the interconnections between the human, animal, plant and environmental health sectors, including biosecurity (e.g., as outlined in the One Biosecurity framework among others) *(established but incomplete)* {1.6.7.2, 6.3.1, 6.7.2.2}.

D27 National-scale strategies and action plans are instrumental to successfully managing biological invasions as part of a context-specific integrated governance approach *(well established)* {6.2.3.2, 6.3.2.1, 6.7.2.3}. The national strategies and action plans could be developed or updated to align with and implement the Kunming-Montreal Global Biodiversity Framework, particularly Target 6, as well as other relevant international guidelines for sustainable development, through aspirational, ambitious and realistic approaches *(well established)* {6.1.2, 6.2.3.2, 6.3.2.1, 6.6.3, 6.7.2.3}. Coordinated efforts to strengthen national regulatory instruments, including for the regulation of online trade {6.3.1.4(3)}, are key to reducing the transport and introduction of invasive alien species *(established but incomplete)* {6.3.1.1, 6.7.2.1}. Voluntary codes of conduct **(Box SPM.1)** have limitations but they can be a valuable part of integrated systems to reduce the risk of biological invasions, when in line with relevant international obligations and national legislations *(established but incomplete)* {6.3.1.4(4)}. Adequately designed and implemented national biodiversity strategies and action plans are instruments to help manage biological invasions and mitigate the impacts of invasive alien species *(established but incomplete)* {6.1.2, 6.3.3.1}. Implementation of strategies could be accelerated by measuring and monitoring resourcing of actions, implementation processes, outputs and outcomes of policy management *(established but incomplete)* {Table 6.5, Box 6.3, 6.6.3}, which could also create a conducive policy environment for the utilization of environmentally sound technologies *(established but incomplete)* {6.3.3.4}.

D28 Long-term commitment and resourcing from governments and institutions will support the implementation of strategic actions to underpin the integrated governance of biological invasions *(established but incomplete)* {6.2.3.2, 6.5.1, 6.5.3, 6.5.7}. With adequate levels of sustained investment and resources **(Table SPM.2)**, including support to developing countries {6.5.7}, specific options that address the gaps and inconsistencies in current policy instruments and coordination can be implemented over appropriate timeframes *(established but incomplete)* {6.7.2.2, 6.7.2.3}. Regulatory and market-based instruments such as tax relief and subsidization can be used to incentivize action on and investment in prevention and control of invasive alien species *(established but incomplete)* {6.3.1, 6.5.1, 6.5.2}, especially when responsibility for the burden of biological invasions, including environmental liability, is shared **(Figure SPM.7)**. These instruments may be non-market mechanisms or voluntary codes of conduct **(Box SPM.1)** {6.3.1.4}, transparent and conducive regulatory settings for new technologies {6.3.3.4, 6.7.2.7}, information-sharing {6.6.2, 6.7.3}, product labelling {6.3.1.4} or direct regulatory intervention {6.3.3.1, 6.3.3.3}. Regulations could be enforced with economic penalties and tariffs *(established but incomplete)* {6.5.1, 6.5.2}. However, taxation incentives, international standards and cost-sharing mechanisms are generally the preferable policy instruments for encouraging entities to participate in prevention and control activities *(established but incomplete)* {5.6.2.1, 6.5.1, 6.5.2, 6.5.4, 6.5.5, 6.5.6}. Efforts to overcome the asymmetries and differences in resource capacity among stakeholders and the potential unequal burden and responsibilities of addressing the causes and impacts of invasive alien species can be embedded in policies *(established but incomplete)* {6.2.3.3, 6.4.4.3}. Cost-benefit and "willingness to pay" analyses and stakeholder consultation can support the development of national policies to assist in justifying the use of public resources and developing the most appropriate incentives *(established but incomplete)* {5.2.2.1.i, 6.2.3.1(2), 6.2.3.4}.

D29 Public awareness and engagement contribute to the effective management of biological invasions *(well established)* {5.6.2.1, 6.2.2(9), 6.3.1.4, 6.4.1, 6.6.2.1, 6.7}. Public understanding of the risks associated with invasive alien species is particularly important for preventing new introductions *(well established)* {6.2.2(9), 6.4.1}. Increased understanding of possible biological invasions and the negative impacts of invasive alien species can be achieved through public awareness campaigns {Box 6.11, 6.7.2.5}, education across all age groups {6.7.2.4} and citizen science *(established but incomplete)* {5.4.2.2.a, 6.6.2.1}. Engagement of the general public *via* citizen science platforms, awareness campaigns and community-driven eradication campaigns also contributes to establishing shared responsibilities for managing biological invasions *(established but incomplete)* {6.7.2.5}. Surveillance for detecting invasive alien species through citizen science and social media provides broader security by empowering and engaging the public *(established but incomplete)* {5.4.2.1.a, 5.4.2.2.a, 6.6.2.1}. Communication is an effective tool for inspiring collective action to monitor and control invasive alien species {6.2.3.1(4), 6.2.3.4, 6.4.4.4} by supporting the co-design of management actions, knowledge exchange and enhanced partnerships among stakeholders and researchers *(established but incomplete)* {6.2.3.3, 6.4.4.3}. It can also enable alignment of resource managers' responses with national plans and policy priorities *(well established)* {6.3.1.3, 6.3.2.1}. An effective communications strategy considers the most appropriate timing, media and channels/interfaces for the target audience *(established but incomplete)* {Box 6.13, 6.6.2.6}.

D30 Indigenous Peoples and local communities have invaluable knowledge systems that could contribute to addressing biological invasions *(established but incomplete)* {Box 4.18, 5.5.3, 5.5.4, 6.4.3.2}, yet their lack of land tenure and access rights can limit the extent to which they are able to take action *(well established)* {3.2.5, 6.4.3.1}. Indigenous Peoples and local communities can be partners in co-developing policies and strategies to address biological invasions while giving consideration to the challenge of conflicting perceptions and values in order to achieve consensus on management actions *(established but incomplete)* {5.6.1.2, 6.2.3.3, 6.4.3.1}. Participation of Indigenous Peoples and local communities can be enhanced with sufficient legal, political and financial support *(well established)* {6.4.3, Box 6.16}. Successful strategies respect the knowledge, priorities and rights of Indigenous Peoples and local communities, including customary governance systems, in accordance with national legislation *(established but incomplete)* {5.1.3, 5.2.1, 5.6.2, 6.4.3}. In cases where the impact of invasive alien species on the quality of life of Indigenous Peoples and local communities is unavoidable, those communities need ongoing support and

Table SPM 2 **Options for strengthening the governance of biological invasions at national, regional and global scales.**

Indication of the duration of investment needed to implement different options. The contribution of each of these options, together forming integrated governance, are given in **Figure SPM.7**. This table presents concrete options for action.

Governance purpose	Options	Duration of investment needed
Coordination and resourcing	Enhance multilateral coordination and collaboration to support the integrated governance of biological invasions	Ongoing
	Engage broadly across affected and responsible parties	Ongoing
	Build capacity to enable strategic actions	Ongoing
Policy	Share efforts, commitments and understanding of the specific roles of all	Short
	Strengthen compatibility of relevant regulatory instruments	Periodic
	Use national strategy and planning for invasive alien species to achieve policy implementation	Periodic
	Support, fund and mobilize resources for innovation, research and environmentally sound technology	Ongoing
	Support information systems, infrastructures and open and equitable access to information on invasive alien species	Short
Research, information, and technology	Invest in information systems for invasive alien species for information-sharing within and across countries	Ongoing
	Maintain up-to-date information on necessary and enabling indicators	Ongoing
	Monitor policy and management effectiveness and resourcing levels	Ongoing
	Develop new solutions through research and technology	Ongoing

Short Periodic Ongoing

adequate resources to respond to the challenges of living with invasive alien species *(established but incomplete)* {1.6.7.2, 6.2.3.2, 6.2.3.5}.

D31 Open and interoperable information systems, supported by international cooperation, play a critical role in tackling biological invasions *(established but incomplete)* {6.2.3.1(3), 6.6.2.2, 6.7.2.6}. Strengthening existing open information systems can facilitate the management of biological invasions, including prioritization of actions, early detection and rapid response, and can improve the effectiveness of regulations *(established but incomplete)* {5.4.1, 6.6.2.3}. Open information systems can substantially reduce the costs of management by ensuring targeted and appropriate responses, avoiding duplication of efforts and facilitating the evaluation of the effectiveness of policy instruments using indicators **(Table SPM.2)** *(well established)* {6.6.2.4, 6.6.2.6, 6.6.3}. The "rate of invasive alien species establishment" headline indicator adopted for monitoring progress towards Target 6 of the Kunming-Montreal Global Biodiversity Framework provides opportunities to build on existing indicators of biological invasions **(Table SPM.A1)** {6.6.3}. Collaboration and networking among stakeholders and governments can ensure equitable knowledge access *(established but incomplete)* {6.2.3.3, 6.2.3.4} and improve understanding of the context-specific features of biological invasions. It can also improve the availability of data and knowledge across geographic regions, habitats and taxonomic groups and reduce the wide variation in response capability *(established but incomplete)* {6.2.3.3, 6.4.1, 6.7.2.6}. Through citizen science, information systems have the potential to engage people, raise awareness and increase the availability of data *(established but incomplete)* {6.6.2.1}.

D32 Existing evidence of the magnitude and extent of the impacts of invasive alien species supports immediate, strategic and sustained action to successfully address biological invasions *(well established)* {1.1, 2.2, 3.6.3, 4.3.1, 4.4.1, 4.5.1, 5.6.2.5, 6.7.2}. The available data and knowledge reviewed for this assessment vary across regions, units of analyses, taxonomic groups and time because of language barriers, lack of targeted policies and legislation, lack of resources, uneven research capacity, data accessibility and other factors **(Table SPM.A1)**, contributing to gaps in data and knowledge *(well established)* {2.7, 3.6.1, Box 3.12, Box 3.13, 4.7.2, 6.6, Table 6.10}. Nonetheless, filling knowledge and data gaps, particularly at local scales, can bring about important improvements in the cost-effectiveness and success of prevention and management actions *(well established)* {6.6.1, 6.6.2}. For example, it would be particularly beneficial to increase the availability of information on invasive alien invertebrates and microorganisms; improve knowledge of the impacts of invasive alien species in parts of Africa, Central Asia and Latin America; gain a better understanding of the role of indirect and interacting drivers; develop management options for invasive microorganisms and marine species; and establish the effectiveness of different policy instruments *(established but incomplete)* (see **Table SPM.A1** for a comprehensive presentation of knowledge gaps). Enhancing research capacity in some regions and collaboration between biological invasion experts in the developed and developing world and across knowledge systems could improve data and information availability as well as understanding of the context-specific features of invasive alien species and their impacts *(established but incomplete)* {6.2.4, 6.6.1.1(3)}. With political will, strategic long-term commitment and sufficient resources, management of biological invasions is an attainable goal *(well established)* {Boxes 5.2, 5.4, 5.5, 5.6, 5.7, 5.8, 5.9, 5.11, 5.12, 5.14, 5.15, 5.16, 5.17, 5.19, 5.21, 6.7.3}.

D33 Successfully addressing biological invasions can also strengthen the effectiveness of policies designed to respond to other drivers *(established but incomplete)* {5.6.1.3, 6.3, 6.7.2.2}. Mitigating the risks of invasive alien species will contribute to the effective delivery of the 2030 Agenda for Sustainable Development, including the Sustainable Development Goals, especially those addressing the conservation of marine (Goal 14) and terrestrial biodiversity (Goal 15 including, but not restricted to, Target 15.8), food security (Goal 2), sustainable economic growth (Goal 8), sustainable cities (Goal 11), climate change (Goal 13), and good health and well-being (Goal 3) *(established but incomplete)* {6.7}. An integrated governance approach that acknowledges the interactions between invasive alien species and other drivers, including climate change, direct exploitation of natural resources, pollution and land- and sea-use, alongside human, animal and plant health, can identify where to best direct policy alignment and mutually supportive efforts *(established but incomplete)* {3.1.5, 6.2.4, 6.7.2.1, 6.7.2.2, 6.7.2.5}. Evidence-based policy planning can reflect the interconnectedness of the drivers so that efforts to solve one problem do not exacerbate the magnitude of others and may even have multiple benefits *(established but incomplete)* {3.2.5, Box 3.9, 5.6.1.3, 6.2.4, 6.3.1.1(1), 6.7.2.2}.

APPENDICES

APPENDIX 1
Communication of the degree of confidence

Figure SPM A 1 **The four-box model for the qualitative communication of confidence.**

Confidence increases towards the top-right corner, as suggested by the increasing strength of shading. Source: IPBES (2016).[12] Further details of the approach are documented in the *IPBES Guide on the Production of Assessments*.[13]

In this assessment, the degree of confidence in each main finding is based on the quantity and quality of evidence and the level of agreement regarding that evidence **(Figure SPM.A1)**.

The evidence includes data, theory, models and expert judgement.

- **Well established:** there is a comprehensive meta-analysis or other synthesis or multiple independent studies that agree.
- **Established but incomplete:** there is general agreement, although only a limited number of studies exist; there is no comprehensive synthesis, and/or the studies that exist address the question imprecisely.
- **Unresolved:** multiple independent studies exist but their conclusions do not agree.
- **Inconclusive:** there is limited evidence and a recognition of major knowledge gaps.

12. IPBES (2016): *Summary for Policymakers of the Assessment Report on Pollinators, Pollination and Food Production of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Potts, S.G., Imperatriz-Fonseca, V. L., Ngo, H. T., Biesmeijer, J. C., Breeze, T. D., Dicks, L. V., Garibaldi, L. A., Hill, R., Settele, J., Vanbergen, A. J., Aizen, M. A., Cunningham, S. A., Eardley, C., Freitas, B. M., Gallai, N., Kevan, P. G., Kovács-Hostyánszki, A., Kwapong, P. K., Li, J., Li, X., Martins, D.J., Nates-Parra, G., Pettis, J.S., Rader, R. and Viana, B.F. (eds.). IPBES secretariat, Bonn, Germany. http://doi.org/10.5281/zenodo.2616458.
13. IPBES (2018): *IPBES Guide on the Production of Assessments. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*, Bonn, Germany. Available at: https://ipbes.net/guide-production-assessments.

APPENDIX 2
Synthesis of knowledge and data gaps

Table SPM A 1 **Table of knowledge and data gaps**

Synthesis of the most important knowledge and data gaps identified and collated through the assessment. Confidence levels in the summary for policymakers were allocated with full consideration of the gaps listed in the table; closing those gaps would strengthen the understanding of biological invasions. Experts have assessed the estimated cost and scientific challenge of closing these gaps, as well as the potential gain in increasing understanding of and successfully tackling biological invasions globally (from very low to very high). The listed gaps may not be relevant at local or regional scales.

CATEGORY	GAP	IMPLEMENTATION CHALLENGE		POTENTIAL GAIN	
		Estimated research cost	Estimated scientific challenge	For taking management action	For better understanding biological invasions
Gaps in biomes, units of analysis and species groups	Incomplete or lack of inventories of invasive alien species in marine, tropical and Arctic ecosystems {2.5.2.1, 2.5.2.4, 2.5.2.5, 2.5.4}	🟠	🟣	🟣	🟣
	Incomplete or lack of inventories of invasive alien microorganisms and invertebrates {2.3.1.11, 2.3.3.3}	🟣	🟣	🟣	🟣
	Lack of understanding of the drivers of change that facilitate biological invasion for some animal groups (notably invertebrates), fungi and microbes {3.6.1}	🟠	🟠	🟣	🟣
	Lack of understanding and synthesis of the impacts of invasive alien microbes {4.7.2}	🟠	🟠	🟣	🟣
	Poor understanding of drivers of change that facilitate biological invasions in aquatic and marine systems {3.6.1}	🟠	🟠	🟣	🟣
	Lack of data on successful restoration attempts in terrestrial and marine systems {5.5.6, 5.6.2.1}	🔵	🔵	🟣	🔵
Regional gaps in data and knowledge	Comparatively incomplete inventories of invasive alien species in Africa and Central Asia {2.4.2.5, 2.4.5.5}	🟠	🟣	🟣	🟣
	Comparative lack of understanding of the drivers of change that facilitate biological invasions in developing economies {Box 3.12}	🟠	🟠	🟣	🟣
	Lack of data and knowledge of the drivers of biological invasions in sub-Saharan Africa, tropical Asia and South America {3.6.1}	🟣	🟠	🟣	🟣
	Incomplete data on the impacts of invasive alien species across Africa and Central Asia {4.7.2}	🟠	🔵	🟣	🟣
Interoperable data for monitoring invasive alien species and effects of drivers of biodiversity change	Lack of standardization of terminology for invasive alien species monitoring {2.4.4.5, 6.6.2.3, 6.6.2.7}	🟢	🟢	🟠	🟠
	Lack of information on the role of indirect drivers, especially governance and sociocultural drivers, in affecting biological invasions {3.1.5, 3.6.1, Box 3.13}	🔵	🔵	🟣	🟣
	Lack of understanding of the net effects of multiple interacting drivers in shaping and promoting biological invasions {3.5, Box 3.10, 3.6.1, Box 3.13}	🟢	🟠	🟣	🟣
	Lack of knowledge on interactions and feedback across drivers in promoting invasions {3.1.5, 3.6.1}	🔵	🟠	🟣	🟣

CATEGORY	GAP	IMPLEMENTATION CHALLENGE		POTENTIAL GAIN	
		Estimated research cost	Estimated scientific challenge	For taking management action	For better understanding biological invasions
Interoperable data for monitoring invasive alien species and effects of drivers of biodiversity change	Lack of integration of impact data and knowledge sources across languages {4.7.2}	●	●	●	●
	Incomplete data to undertake risk management, cost-effective species-based surveillance and detection of fungi, microbes and marine pests {Table 5.11}	●	●	●	●
	Incomplete data to prioritize biological invasion management under climate, sea- and land-use change {5.6.1.3}	●	●	●	●
	Lack of inventories at fine scales and for specific taxon and biome contexts to support decision-makers in determining when to implement species-based or site-based management (or both) {5.6.2.1, 5.7}	●	●	●	●
	Incomplete data to develop pathway risk assessments and management for different taxonomic groups and biomes {Table 5.11, 5.6.2.5}	●	●	●	●
	Incomplete data and understanding of site-based and ecosystem-based management concepts {5.6.2.1}	●	●	●	●
	Incomplete data and understanding of the conditions that facilitate successful integration of policy developments into management plans {6.6.1.4}	●	●	●	●
	Lack of indicators of the various dimensions of biological invasion that are policy-relevant, sensitive, reliable, relevant at national and global scales, sustained for medium-to-long-term tracking of progress and part of a responsive policy environment {6.6.3}	●	●	●	●
Gaps in how invasive alien species affect nature's contributions to people	Incomplete data on impacts on nature's contributions to people and good quality of life {4.7.2}	●	●	●	●
Management and policy approaches	Lack of control options for marine invasive alien species and invasive alien microbial fungal pathogens of plants and animals {5.6.1.1}	●	●	●	●
	Lack of agreed-upon methods of supporting management decision-making for invasive alien species with both positive and negative impacts {5.6.1.2}	●	●	●	●
	Lack of methods of managing pathways for invasive alien species arriving as contaminants, or through shipping containers, e-commerce (legal/illegal), biofouling or ports, and across land borders and along trade supply chains {Table 5.11, 5.6.2.4}	●	●	●	●
	Lack of methods for adaptive management of invasive alien invertebrates and plants using alternative approaches given the declining number of chemical control options {5.6.2.5}	●	●	●	●
	Lack of eradication guidelines and strategies for generalist invasive alien invertebrates, diseases and hard-to-detect freshwater and marine invasive alien species {5.6.2.1, Table 5.11}	●	●	●	●
	Lack of scenarios and models of invasive alien species that consider interactions with other drivers of global change {2.6.5, 6.6.1.6}	●	●	●	●
	Missing information on the implementation of adaptive-collaborative governance for biological invasions and factors important to the success of that governance strategy {6.4.4.5}	●	●	●	●
	Incomplete data on the effectiveness of policies, management strategies and actions related to biological invasions {6.1.3, 6.6.3}	●	●	●	●

CATEGORY	GAP	IMPLEMENTATION CHALLENGE		POTENTIAL GAIN	
		Estimated research cost	Estimated scientific challenge	For taking management action	For better understanding biological invasions
Gaps to fill to support the implementation of policy and management	Lack of tools and frameworks to predict biological invasions {6.2.1, 6.6.1.6, 6.7.2.7}	Intermediate	Intermediate	High	High
	Lack of tools to reduce the barriers to information-sharing within and across countries {6.6.2}	Intermediate	Intermediate	High	Very high
	Lack of research and data on how best to implement integrated governance systems to manage biological invasions {6.6.1.3, 6.6.1.4, 6.6.2}	Intermediate	Intermediate	Very high	High
	Design principles for an integrated governance system to manage biological invasions {6.7.2.3, 6.7.3}	Intermediate	High	High	Intermediate
	Lack of mechanisms that allow effective collaboration among different elements of the socioecological systems {Figure 6.7, 6.7}	Intermediate	Intermediate	High	Intermediate
Gaps in knowledge on invasive alien species of particular relevance to Indigenous Peoples and local communities	Lack of information on invasive alien species status and trends on land and water managed by Indigenous Peoples and local communities {Box 2.6}	High	High	High	High
	Lack of information on Indigenous and local knowledge, values and culture regarding the drivers and impacts of invasive alien species on land and water managed by Indigenous Peoples and local communities {1.6.7.1, Box 3.12}	High	High	Very high	Very high
	Lack of understanding of and mechanisms for sharing knowledge on invasive alien species and their drivers, impacts, management and governance among Indigenous Peoples and local communities and researchers and other outsiders {6.6.1.5}	High	High	Very high	Very high
	Lack of consideration of the knowledge and perceptions of Indigenous Peoples and local communities in scenarios and models {1.6.7.3, 4.7.1, 6.6.1.6}	High	High	High	Intermediate

Very low · Low · Intermediate · High · Very high

[a] A headline indicator has been adopted for planning and tracking of progress towards Target 6 of the Kunming-Montreal Global Biodiversity Framework, with opportunities to build on existing indicators for biological invasions {6.6.3}.

APPENDIX 3
Examples of data and knowledge products

Information components including description and importance of the information for documenting and managing biological invasions of existing invasive alien species databases that may provide relevant information.

Websites are provided at the first mention of each database (see chapter 2 for databases relevant for status and trends and chapter 6, section 6.6.3 for databases supporting policy options). Gaps identified within the data and knowledge products are also given {Table 5.4}.

Fields	Description	Database purpose	Examples of data and knowledge products	Identified gaps
Taxonomy	Scientific name, higher taxonomy, synonyms, common names	Name consistency and locating specimens	• GBIF – https://www.gbif.org/ • World Register of Introduced Marine Species – http://www.marinespecies.org/introduced/ • FishBase – https://fishbase.org/ • Plant List – http://www.theplantlist.org/ • The Reptile Database – http://www.reptile-database.org/ • AlgaeBase – https://www.algaebase.org/ • IUCN Red List of Threatened Species – https://www.iucnredlist.org/	Underrepresented biomes and taxa
Identification	Identification guides, diagnostic tools	Correct identification, early detection	• iNaturalist – https://www.inaturalist.org • Lucidcentral – https://www.lucidcentral.org • Antweb – a comprehensive diagnostic tool for ants – http://antweb.org/ • Plant net – https://plantnet.rbgsyd.nsw.gov.au/ • eBird – https://ebird.org/home • BioNET – EAFRINET – https://keys.lucidcentral.org/keys/v3/eafrinet/plants.htm • Portaleei Latin America – http://portaleei.fcien.edu.uy/	
Ecology	Including habitat, species interactions (e.g., host species)	Management risk assessment	• Global Invasive Species Database (GISD) – http://www.iucngisd.org/gisd • Centre for Agriculture and Bioscience International Invasive Species Compendium – https://www.cabi.org/isc • FishBase • National invasive alien species databases – http://www.inbiar.uns.edu.ar/; http://bd.institutohorus.org.br; https://caribbeaninvasives.org; https://sieei.udelar.edu.uy; https://guyra.org.py; https://invasoras.biodiversidad.gob.ec	
Spatial data	Distribution, native and introduced range, occurrence	Origin, management, risk assessment	• Global Invasive Species Database • Global Register of Introduced and Invasive Species (GRIIS) – http://www.griis.org/ (Pagad *et al.*, 2018, 2022b, 2022a) {Table 5.4} • Centre for Agriculture and Bioscience International Invasive Species Compendium • FishBase • Global Naturalized Alien Flora (GloNAF) – https://glonaf.org	

Fields	Description	Database purpose	Examples of data and knowledge products	Identified gaps
Spatial data	Distribution, native and introduced range, occurrence	Origin, Management, Risk assessment	• Global Avian Invasions Atlas – https://doi.org/10.6084/m9.figshare.4234850.v1 • SeaLifeBase – https://www.sealifebase.ca • WOAH – https://www.woah.org/en/what-we-do/animal-health-and-welfare/disease-data-collection/world-animal-health-information-system/ • European Alien Species Information Network – https://easin.jrc.ec.europa.eu/easin/# • Pacific Islands Ecosystems at Risk – http://www.hear.org/pier/ • Species observations for the United States and Territories – https://www.gbif.us • Atlas of Living Australia. Analytic software platforms, extensive and open source – www.ala.org.au • National invasive alien species databases • Biomodelos – Biomodels of potential distribution maps and invasive species fauna and flora in Colombia – http://biomodelos.humboldt.org.co/en • International Union for Conservation of Nature Red List of Threatened Species • Regional plant protection organizations – https://www.ippc.int/en/external-cooperation/regional-plant-protection-organizations/	
Status and provenance	Biological invasion status in introduced range including abundance, occurrence (extent of spread) and invasiveness	Origin, prioritization and management prioritization	• Global Invasive Species Database • Global Register of Introduced and Invasive Species • Centre for Agriculture and Bioscience International Invasive Species Compendium • FishBase • European Alien Species Information Network • Pacific Islands Ecosystems at Risk • World Register of Introduced Marine Species • SeaLifeBase – https://www.sealifebase.ca/ • WOAH World Animal Health Information System – disease status • National invasive alien species databases	
Primary and secondary pathways	Intentional or unintentional pathways of introduction and spread	Biosecurity management	• Global Invasive Species Database • Global Register of Introduced and Invasive Species • Centre for Agriculture and Bioscience International Invasive Species Compendium • FishBase • European Alien Species Information Network • Pacific Islands Ecosystems at Risk • World Register of Introduced Marine Species • Database on Introductions of Aquatic Species • IPPC Documentation on ISPM – https://www.ippc.int/en/core-activities/standards-setting/ispms/ • National invasive alien species databases – http://www.inbiar.uns.edu.ar/	Secondary pathways classification inconsistent or missing
Monitoring and surveillance	Data from multiple sources in a real time	Early detection	• Early Detection and Distribution Mapping System – https://www.eddmaps.org/	
Impact	Environmental and socio-economic impacts, mechanisms of impact, outcomes of these impacts and ecosystem services impacted	Risk assessment policy management	• Global Invasive Species Database • Global Register of Introduced and Invasive Species • Centre for Agriculture and Bioscience International Invasive Species Compendium	No transparent, standardized way to report on impacts

Fields	Description	Database purpose	Examples of data and knowledge products	Identified gaps
Impact	Environmental and socio-economic impacts, mechanisms of impact, outcomes of these impacts and ecosystem services impacted	Risk assessment policy management	• InvaCost database – https://figshare.com/articles/dataset/InvaCost_References_and_description_of_economic_cost_estimates_associated_with_biological_invasions_worldwide_/12668570/4 • Millennium ecosystem assessment – https://www.millenniumassessment.org • IUCN Red List of Threatened Species – https://www.iucnredlist.org/resources/threat-classification-scheme • FishBase	No transparent, standardized way to report on impacts
Risk assessments	Developed risk assessments with outcomes	Management	• Global Invasive Species Database • Pacific Islands Ecosystems at Risk • Environmental Impact Classification of Alien Taxa and the Socio-Economic Impact Classification for Alien Taxa • Global Compendium of Weeds – http://www.hear.org/gcw/ • East and South European Network for Invasive Alien Species – www.esenias.org • Pacific Invasive Ants Toolkit – http://www.piat.org.nz/ • National invasive alien species databases	
Policy response	Legislations enacted, regulations, voluntary codes of conduct	Policy management	• ECOLEX – https://www.ecolex.org • FAOLEX – fao.org/faolex/en/ • InforMEA – United Nations Information Portal on Multilateral Agreements – https://www.informea.org • EU Regulations – https://ec.europa.eu/environment/nature/invasivealien/index_en.htm	Databases not searchable for invasive alien species
Eradication	Successes	Management	• DIISE – http://diise.islandconservation.org/ • Global Eradication and Response Database – http://b3.net.nz/gerda/ • National invasive alien species databases	
Control	Management practices, failure, best practices, biocontrol	Management	• Pacific Islands Ecosystems at Risk • Database of introductions of insect biological control agents for the control of insect pests (Cock *et al.,* 2016) {Table 5.4} • Biological Control of Weeds. A world catalogue of agents and their target weeds – https://www.ibiocontrol.org/ • iMapInvasives – sharing information for strategic management – https://www.imapinvasives.org • Centre for Agriculture and Bioscience International Invasive Species Compendium • Pacific Invasive Ant Toolkit • Caribbean Invasive Alien Species Network – https://caribbeaninvasives.org/ • Database of Island Invasive Species Eradications • Global Eradication and Response Database • Early Detection and Distribution Mapping System • East and South European Network for Invasive Alien Species • National invasive alien species databases	No standardized way to report on management outcomes

Chapter **1**

INTRODUCING BIOLOGICAL INVASIONS AND THE IPBES THEMATIC ASSESSMENT OF INVASIVE ALIEN SPECIES AND THEIR CONTROL[1]

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Chapter 1

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Chapter **1**

INTRODUCING BIOLOGICAL INVASIONS AND THE IPBES THEMATIC ASSESSMENT OF INVASIVE ALIEN SPECIES AND THEIR CONTROL

1.1 INTRODUCTION: THE IPBES THEMATIC ASSESSMENT OF INVASIVE ALIEN SPECIES AND THEIR CONTROL

Invasive alien species **(Figure 1.1)**, through the process of biological invasion, are widely recognized as a major threat to nature and nature's contributions to people, with important implications for good quality of life (IPBES, 2018e, 2018f, 2018g, 2018h, 2019; **Glossary**). Biological invasions are a consequence of human activities and invasive alien species are acknowledged as one of the major drivers of local species extinctions within terrestrial and inland water ecosystems (Bellard *et al.*, 2016; IPBES, 2019a; **Chapters 3** and **4**); they have dramatically altered habitats within terrestrial, marine, and freshwater ecosystems around the world (Cacabelos *et al.*, 2020; Liu *et al.*, 2020; **Chapter 4**; **Glossary**). Invasive alien species, alongside other drivers of change in nature, are considered to be one characteristic of a new epoch – the Anthropocene (Capinha *et al.*, 2015; Crutzen & Stoermer, 2000). While the problems associated with invasive alien species have increased over the past century (**Chapters 2** and **4**), considerable progress has been made toward understanding (**Chapters 2**, **3** and **4**) and developing strategies and actions to manage them (**Figure 1.2**; **Chapter 5**). The thematic assessment report on invasive alien species and their control of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES; hereafter termed the IPBES invasive alien species assessment) provides a timely synthesis of this complex but fascinating multidisciplinary field of research to underpin potential options for policy- and decision-making (**Chapter 6**).

Throughout the IPBES invasive alien species assessment, the term biological invasion is used to describe a process involving the transport of a native species outside of its natural range, intentionally or unintentionally, by human activities to new regions where it may become established and spread (Richardson *et al.*, 2010). The term invasive alien species refers to particular species within the context of the process of biological invasion; namely those that negatively impact (**Glossary**) nature and also, in some cases, nature's contributions to people, and good quality of life.

The rapidly growing threat that invasive alien species pose to nature, nature's contributions to people, and good quality of life remains underestimated and, in some cases, overlooked by policy and decision makers (IPBES, 2018a, 2019). However, concerns over the adverse impacts of invasive alien species have driven multiple efforts to establish regional and international initiatives (**Figure 1.2**; Clout & De Poorter, 2005) and policy goals **(Box 1.1)**. A pioneering initiative was the Scientific Committee on Problems of the Environment (SCOPE), which engaged scientists to document biological invasions and invasive alien species from a global perspective in 1982 (J. A. Drake *et al.*, 1989; Mooney *et al.*, 2005).

The overarching aim of the IPBES invasive alien species assessment is to critically evaluate available evidence on the severity of the threat of invasive alien species to inform potential options for decision-making. The need for sustained social-ecological (Kull *et al.*, 2018), interdisciplinary (Vaz *et al.*, 2017) and transdisciplinary approaches (Kapitza *et al.*, 2019), which are sensitive to differing knowledge systems, value perceptions and cultural attributes, is acknowledged throughout this assessment and will be critical in addressing the recently adopted goals of the Kunming-Montreal Global Biodiversity Framework (CBD, 2022).

While previous regional, global and thematic IPBES assessments have considered biological invasions and invasive alien species, an in-depth and quantitative and qualitative global analysis of them has not been conducted. Therefore, the IPBES invasive alien species assessment not only extends the findings of the previous IPBES assessments, including the IPBES Global Assessment Report on Biodiversity and Ecosystem Services (IPBES, 2019), but addresses important gaps in information. Ultimately, through the synthesis and harmonization of information at a global scale, the IPBES invasive alien species assessment examines the magnitude of the threat of invasive alien species to nature, nature's contributions to people, and good quality of life **(Box 1.2)**.

The term biological invasion describes the process involving the intentional or unintentional transport or movement of a species outside its natural range by human activities and its introduction to new regions, where it may become established and spread.[1]

Native species (synonym indigenous species) are taxa that have originated in a given area (their natural range) without human involvement, or that have arrived there without intentional or unintentional intervention of humans, from an area in which they are native. This definition excludes products of hybridization involving alien taxa since "human involvement", in this case, includes the introduction of an alien parent.[2] Some native species can spread or undergo rapid population increase and have harmful impacts. Despite their adverse effects, such native species are not considered invasive alien species.[3]

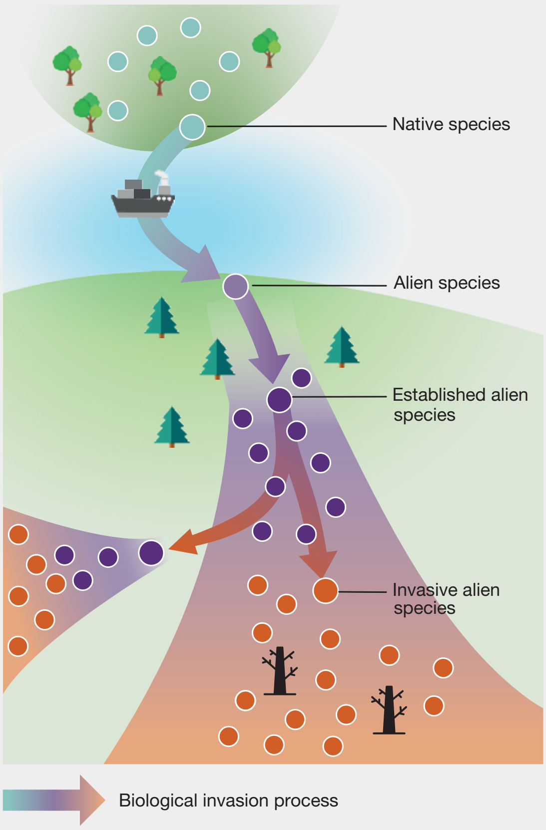
Alien species, as opposed to native species (synonyms exotic, introduced, non-indigenous, non-native), are those whose presence in a region is attributable to human actions, intentional or unintentional, that enable them to overcome biogeographical barriers.[1] Native species that expand their natural range without intentional or unintentional human involvement, for example in response to other anthropogenic drivers such as changes in land use and climate change, are not considered to be alien species.[4, 5] However, a species that spreads to new regions without direct human involvement from a region where it is alien is considered to be alien in the new region.[2]

Established (synonym naturalized) **alien species** produce self-sustaining and viable populations for a given period of time during which climatic extremes typical for the invaded region are experienced, without direct intervention by humans, or despite human intervention.[6, 2, 7]

"**Invasive alien species** are animals, plants or other organisms introduced directly or indirectly by people into places out of their natural range of distribution, where they have become established and dispersed, and generating a negative impact on local ecosystems and species".[8] Invasive alien species are a subset of established alien species that have negative impacts.

Impacts are changes to nature, nature's contributions to people and/or good quality of life.[9] Impacts can be observed or unobserved. Generally, negative impacts become more apparent and problematic when invasive alien species are well established, widespread and present for a long time. Along with their adverse effects, some invasive alien species may have positive impacts providing benefits to some people.

Drivers are factors that directly or indirectly facilitate biological invasions.



1. Richardson *et al.* (2010);
2. Pyšek *et al.* (2004);
3. Wallingford *et al.* (2020);
4. Essl *et al.* (2019);
5. Essl *et al.* (2016);
6. Blackburn, Pyšek *et al.* (2011);
7. Rojas-Sandoval & Acevedo-Rodríguez (2015);
8. IPBES (2018e);
9. Ricciardi *et al.* (2013).

Figure 1 1 **Definitions of important terms used to describe the status of a species from native to invasive alien through the process of biological invasion.**

The definition of native species provides the context for the term natural range. Stages of the biological invasion process (transport, introduction, establishment and spread) are defined in **section 1.3**.

Box 1 1 International policy targets for biological invasions.

The setting of global policy goals and targets is often considered an effective and transparent way to motivate governments and other actors (Kanie & Biermann, 2017). In recent decades, the need for prevention and management (**Glossary**) of biological invasions has been widely recognized by the Conference of the Parties to the Convention on Biological Diversity (CBD), which adopted the Strategic Framework for Biodiversity 2011-2020 in 2010, including the Aichi Biodiversity Targets (United Nations, 1992) which adopted the Strategic Framework for Biodiversity 2011-2020 in 2010, including the Aichi Biodiversity Targets, and the United Nations General Assembly, which adopted the 2030 Agenda for Sustainable Development and its Sustainable Development Goals (SDGs) in 2015. More specifically, two international commitments were made:

> "By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated and measures are in place to manage pathways to prevent their introduction and establishment." Aichi Biodiversity Target 9, Strategic Plan for Biodiversity 2011-2020 (CBD, 2010; **Glossary**).

> "By 2020, introduce measures to prevent the introduction and significantly reduce the impact of invasive alien species on land and water ecosystems and control or eradicate the priority species." Target 15.8, SDG15 (United Nations, 2020a).

These targets were mostly directed towards biodiversity and conservation. However, while the wording of these targets does not address good quality of life directly, they are framed within a broader policy context aimed at conserving biodiversity and ensuring its sustainable use by human communities, the equitable sharing of benefits from genetic resources (CBD, 2020), and the broader goal of achieving a better and more sustainable future for all (United Nations, 2020b). As such the 2020 targets recognized the current and future threats posed by invasive alien species to humanity.

None of the 20 Aichi Biodiversity Targets were achieved at the global level (Secretariat of the CBD, 2020). The Kunming-Montreal Global Biodiversity Framework was adopted in 2022 and includes a target on invasive alien species, Target 6:

> "Eliminate, minimize, reduce and or mitigate the impacts of invasive alien species on biodiversity and ecosystem services by identifying and managing pathways of the introduction of alien species, preventing the introduction and establishment of priority invasive alien species, reducing the rates of introduction and establishment of other known or potential invasive alien species by at least 50 per cent, by 2030, eradicating or controlling invasive alien species especially in priority sites, such as islands." Kunming-Montreal Global Biodiversity Framework (CBD, 2022).

Box 1 2 Overarching questions on biological invasions.

The IPBES invasive alien species assessment addresses 11 overarching questions (IPBES, 2018a).

a. What progress has been made in tackling the Aichi Biodiversity Targets of relevance to invasive alien species globally?

b. What global-level policy initiatives would assist in invasive alien species prevention and management?

c. What are the obstacles to the uptake of invasive alien species prevention and management measures?

d. What methods are available for prioritizing invasive alien species threats?

e. How can networks assist in the prevention and management of invasive alien species? What role can regional partnerships play?

f. Are there perverse policy drivers that unintentionally create risks in relation to invasive alien species?

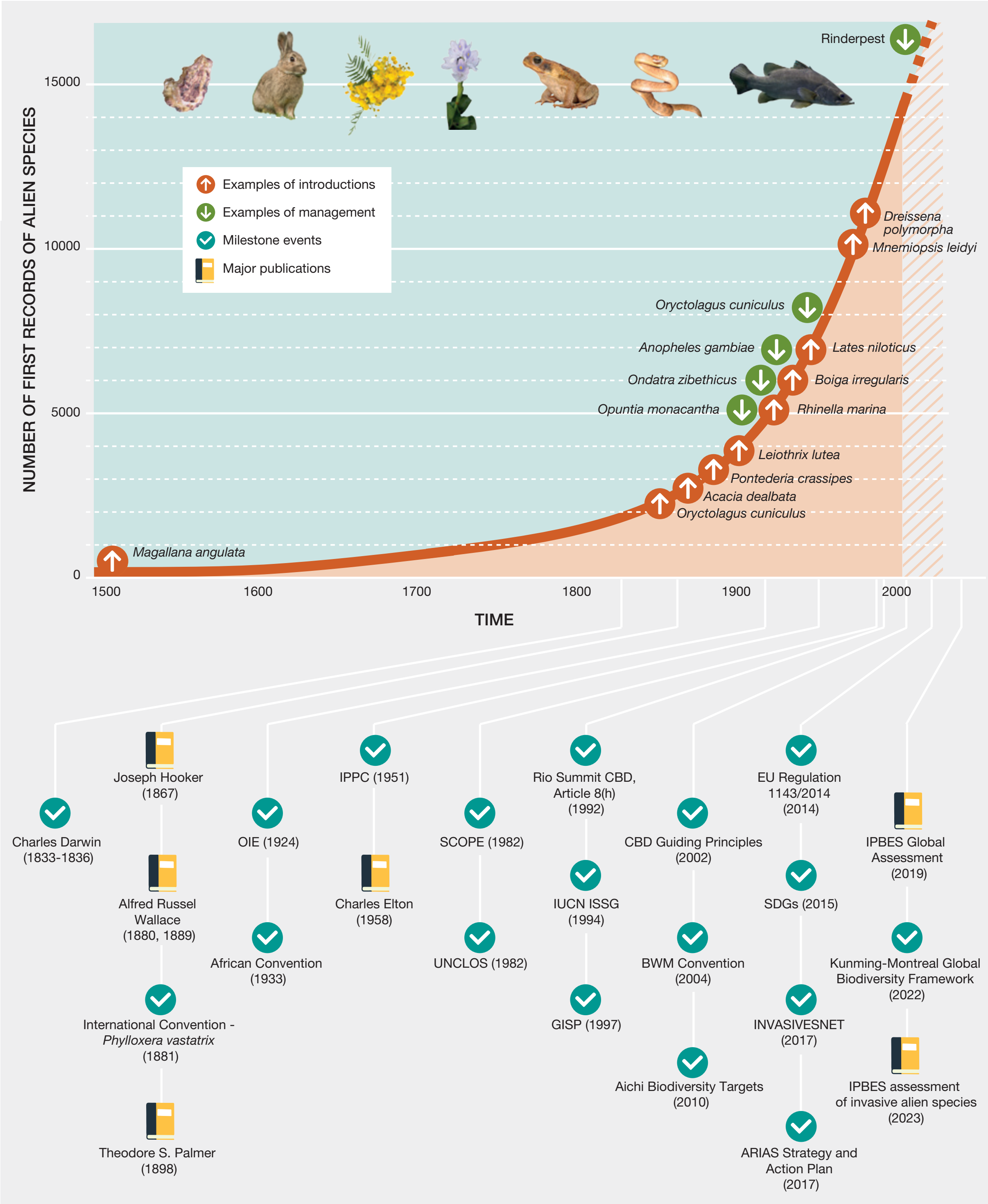
g. How can decision makers decide which issues to tackle first given limited resources?

h. Would there be value in developing a database of effective legislation, monitoring and response systems for invasive alien species, and of those countries and other stakeholders in need of capacity-building?

i. What are the impacts, risks and benefits of invasive alien species for biodiversity and ecosystem services, sustainable development and human well-being?

j. How might policy sectors, businesses, non-governmental organizations and other stakeholders benefit from better prevention and management of invasive alien species?

k. How does one prevent and manage invasive alien species that cause harm to biodiversity but contribute to economic activities?



↑	*Magallana angulata* (Portuguese oyster) was introduced in Europe in the 1500s
↑	*Oryctolagus cuniculus* (rabbit) was first recorded in Australia in 1859
↑	*Acacia dealbata* (acacia bernier) was first recorded in Sri Lanka in 1870
↑	*Pontederia crassipes* (water hyacinth) was first recorded in North America in 1884
↑	*Leiothrix lutea* (red-billed leiothrix) was first introduced in Europe in the late 1800s
↑	*Rhinella marina* (cane toad) was first recorded in Australia in 1935

- *Boiga irregularis* (brown tree snake) was first recorded in Guam in the late 1940s or early 1950s
- *Lates niloticus* (Nile perch) was first recorded in Lake Victoria in 1954
- *Dreissena polymorpha* (zebra mussel) was first recorded in North American Great Lakes in 1986
- *Mnemiopsis leidyi* (sea walnut) was first recorded in the Black Sea in 1982
- Control of *Opuntia monacantha* (common prickly pear) in South Africa (1913) and Australia (1914)
- Eradication of *Ondatra zibethicus* (muskrat) in the United Kingdom in 1939
- *Anopheles gambiae* (African malaria mosquito) was successfully managed in Brazil in the 1930s and early 1940s
- Control of *Oryctolagus cuniculus* (rabbits) in Australia in 1955
- Rinderpest is first wild animal disease to be eliminated globally in 2011
- Charles Darwin observed two European plants invading the pampas, Patagonia (1833-1836)
- International Convention on Measures to be taken against *Phylloxera vastatrix* (1881)
- Creation of the Office International des Epizooties (OIE) in 1924
- African Convention on the Conservation of Nature and Natural Resources: Article 7(5) (1933)
- Adoption of the International Plant Protection Convention (IPPC) in 1951
- Launch of the Scientific Committee on Problems of the Environment (SCOPE) programme on the Ecology of Biological Invasions in 1982
- Adoption of the United Nations Convention on the Law of the Sea (UNCLOS) in 1982
- Opening for signature of the Convention on Biological Diversity (CBD), including Article 8(h) on alien species, in 1992
- Creation of the International Union for Conservation of Nature (IUCN) Invasive Species Specialist Group (ISSG) in 1994
- Launch of the Global Invasive Species Programme (GISP) in 1997
- Adoption of the CBD Guiding Principles annexed to decision VI/23 on alien species, in 2002
- Adoption of the Ballast Water Management Convention (BWM) in 2004
- Adoption of the Strategic Plan for Biodiversity 2011-2020 (including the Aichi Biodiversity Targets) in 2010
- Adoption of the European Union Regulation 1143/2014 on Invasive Alien Species in 2014
- Adoption of the 2030 Agenda for Sustainable Development, including the 17 Sustainable Development Goals (SDGs) in 2015
- Creation of the International Association for Open Knowledge on Invasive Alien Species (INVASIVESNET) in 2017
- Adoption of the Arctic Invasive Alien Species (ARIAS) Strategy and Action Plan in 2017
- Creation of the Global Register of Introduced and Invasive Species (GRIIS) in 2017
- Adoption of the Kunming-Montreal Global biodiversity framework in 2022
- Joseph Hooker – devastation of native plants on islands by introduced plants, goats, and rabbits (1867)
- Alfred Russel Wallace – adverse impacts of introduced plants and animals on continents and islands (1880, 1889)
- Theodore S. Palmer – adverse impacts of introduced birds and mammals including myna in Hawaii (1898)
- Charles Elton – synthesis of evidence across diverse themes to provide first overview of the global scale and escalating adverse impacts of biological invasions (1958)
- IPBES Global Assessment Report on Biodiversity and Ecosystem Services (2019)
- IPBES Thematic assessment of invasive alien species and their control (2023)

Figure 1 2 **Timeline of key strategic events and advances in the understanding of biological invasions.**

There has been considerable progress not only in understanding the process of biological invasions and invasive alien species but also in developing strategies and actions to manage them. The timeline shows milestone events relevant to biological invasions ( ), major publications on biological invasions ( ), examples of invasive alien species' first record ( ), and examples of successful management ( ), with the central line graph illustrating the global escalation in first records of alien species. Data management report available at: https://doi.org/10.5281/zenodo.7560099

1.2 ASSESSMENT STRUCTURE

The first assessment of biological invasions and invasive alien species that is global in scope, the IPBES invasive alien species assessment, is interdisciplinary, spanning environmental and social science as well as the humanities, and comprises six chapters written by experts from all regions of the world.

There are many links and several overarching cross-cutting and key issues across the six chapters **(Figure 1.3)**, but all the chapters can be read as standalone documents presenting syntheses of existing knowledge and highlighting gaps and priorities.

The assessment is composed of six chapters:

- **Chapter 1** introduces the concept of invasive alien species; the risks posed to marine, terrestrial and freshwater ecosystems; the IPBES conceptual framework; the cross-cutting themes (good quality of life, Indigenous and local knowledge, and scenarios and models), and common themes;
- **Chapter 2** assesses past, current and future trends in the spread, pathways, evolutionary change and distribution of invasive alien species;
- **Chapter 3** presents the direct and indirect drivers responsible for the introduction, spread, abundance and dynamics of invasive alien species;
- **Chapter 4** assesses the impacts of invasive alien species on nature and nature's contributions to people and good quality of life;
- **Chapter 5** evaluates the effectiveness of past and current programmes and tools for the global, national and local prevention and management of biological

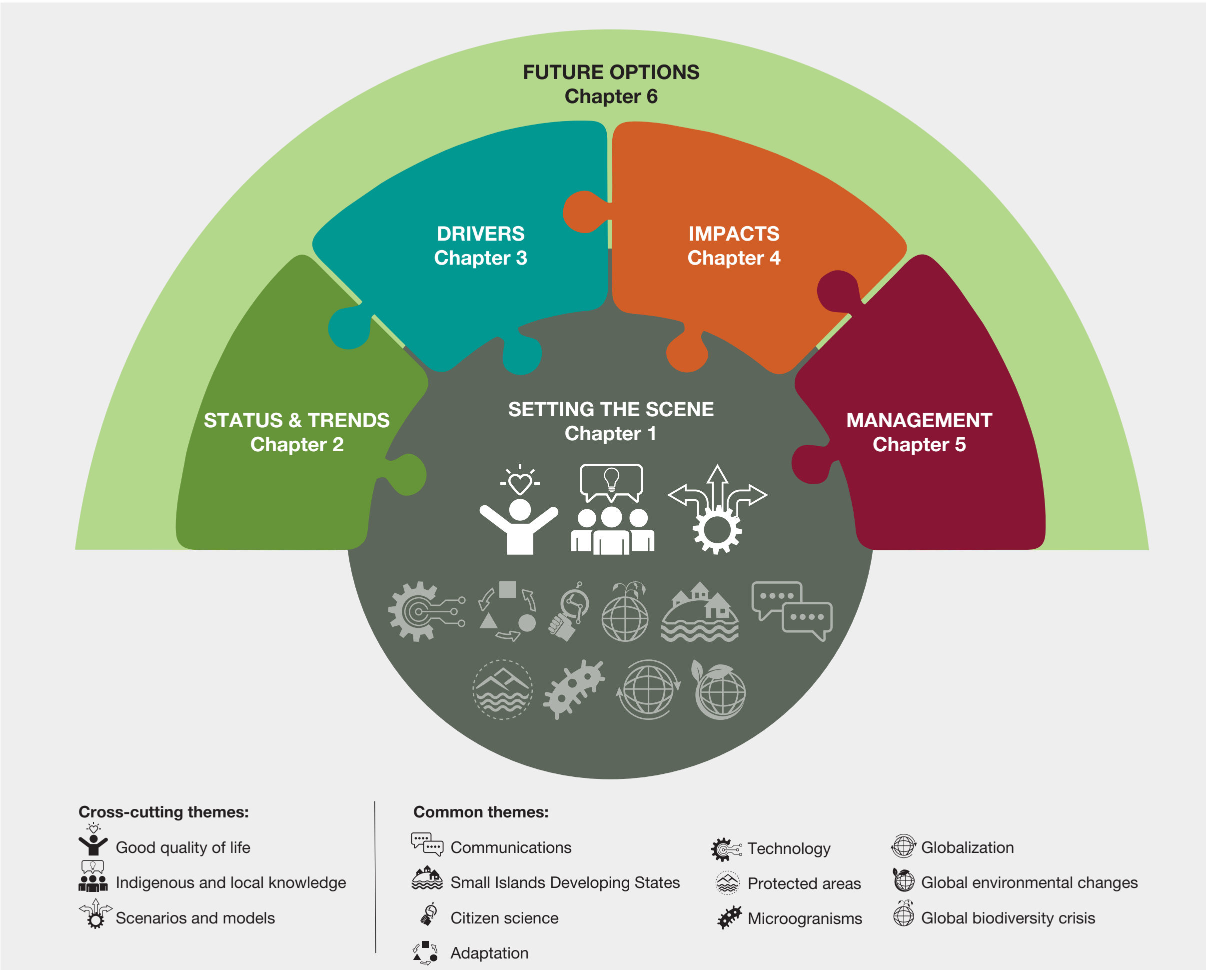


Figure 1 3 **Structure of the IPBES thematic assessment of invasive alien species and their control.**

invasions and invasive alien species and their impacts; and

- **Chapter 6** introduces future options for the prevention and management of biological invasions and invasive alien species and provides an analysis of possible policies and support tools for policy and decision makers.

Three cross-cutting themes – 1) Indigenous and local knowledge systems (**Glossary**), 2) good quality of life including human health, and 3) scenarios and modelling of trends (**Glossary**) and development of robust projections, are featured prominently throughout the IPBES invasive alien species assessment **(Figure 1.3)**. Several key issues, with relevance to two or more of the chapters, emerged during the assessment including globalization, adaptation, environmental change, the global biodiversity crisis, the role of technology, the role of communication, citizen (or community) science, the specific context of Small Island Developing States (SIDS), the role of protected areas (terrestrial, coastal, and marine) and of microorganisms. In many chapters these topics will appear as case studies. As this IPBES assessment will demonstrate, addressing invasive alien species, which are affecting many facets of the socioecological systems in which people live, can have far-reaching benefits for biodiversity and human health, and will shape the ability of future generations to live healthy, sustainable lives.

1.3 INVASIVE ALIEN SPECIES: WHAT THEY ARE AND WHY THEY MATTER

1.3.1 What are invasive alien species?

The term "alien" (synonyms: non-native, exotic, introduced, non-indigenous, allochthonous) species refers to species whose presence in a region is attributable to human actions, intentional or unintentional, that enable them to overcome biogeographical barriers (Essl *et al.*, 2018; Richardson *et al.*, 2010; Rojas-Sandoval & Acevedo-Rodríguez, 2015). It is widely acknowledged that some alien species (i.e., invasive alien species) can become established, spread (dispersed) and cause dramatic biotic and abiotic changes in the ecosystem to which they are introduced, resulting in the reduction in abundance or even extinction of native species, and/or major shifts in ecosystem functioning, and/or major adverse health, economic, social, or cultural impacts on human communities. Invasive alien species are defined in the scoping report for this assessment as "animals, plants or other organisms introduced directly or indirectly by people into places out of their natural range of distribution, where they have become established and dispersed, and generating an impact on local ecosystems and species" (IPBES, 2018e; **Figure 1.1** and **Glossary**). Although much of the focus of this assessment is on the negative impacts of invasive alien species, benefits are also discussed.

Invasive alien species can be introduced unintentionally or intentionally, and as these terms are more commonly used than directly or indirectly, they have been adopted throughout this assessment. Domestic or managed alien animals and plants are not considered to be invasive alien species while they remain in captivity or are managed by humans, but such species that establish feral or wild populations outside of captivity or cultivation would be termed invasive alien species. Furthermore, it is important to note that feral populations of domestic or managed animals (e.g., goats and fish) can have considerable adverse impacts prior to establishing sustained populations in the wild. Native species that expand their natural range without human involvement, for example in response to other anthropogenic drivers including land- and sea-use and climate change, are not considered to be alien species even though some of these range expansions result in dramatic ecosystem-level changes (**Figure 1.1**; Cannone *et al.*, 2022).

Invasive alien species are generally considered problematic because they cause environmental harm and also, in some cases, affect good quality of life. This standpoint is consistent with Article 8(h) of the CBD, which calls on the parties to "prevent the introduction of or control or eradicate those alien species that threaten ecosystems, habitats or species" **(Box 1.1)**. The term "invasive alien species" was adopted by the CBD Guiding Principles for the Prevention, Introduction and Mitigation of Impacts of Alien Species that Threaten Ecosystems, Habitats or Species (CBD, 2002; **Chapter 6**, **Table 6.3**) to define species whose introduction and spread threaten biological diversity. However, perceptions of invasive alien species may vary amongst stakeholders and Indigenous Peoples and local communities (**section 1.5.2**; see also **Chapter 5**, **section 5.6.1.2**) and it is therefore important to view invasive alien species not in isolation but within the context of the socioecological systems they are affecting (**section 1.5.2**).

It can take time for the negative impacts of some alien species to become apparent and so a precautionary approach (**Glossary**) is often adopted when categorizing an alien species as an invasive alien species (Coutts *et al.*, 2018). Generally authors do not consider the inclusion of impact within the definition of biological invasions, and instead their definition is based exclusively on ecological and biogeographical criteria (Blackburn, Pyšek, *et al.*, 2011; Occhipinti-Ambrogi & Galil, 2004; Pyšek *et al.*, 2004; Rojas-Sandoval & Acevedo-Rodríguez, 2015);

many of the datasets collated for alien species follow this approach (Pyšek *et al.*, 2017; 2020). The definition of invasive alien species, supported by the International Union for Conservation of Nature (IUCN), the CBD and the World Trade Organization (WTO), often used in policy discussions, explicitly assumes that invasive alien species cause adverse impacts on nature and also to the economy and good quality of life, including human health (IUCN, 2000). This IPBES invasive alien species assessment follows the definition of invasive alien species outlined within the scoping report (IPBES, 2018a) which includes the concept of impact on local ecosystems and species. Key terms within this definition are provided in **Figure 1.1**.

1.3.2 How many invasive alien species are there?

Patterns in the numbers of established alien species have been documented for all IPBES regions (**Chapter 2**, **section 2.4**; and specifically Bailey *et al.*, 2020; Genovesi *et al.*, 2009; Lambdon *et al.*, 2008; Turbelin *et al.*, 2017) and most taxonomic groups (**Chapter 2**, **section 2.3**; in particular Dawson *et al.*, 2017; Dyer *et al.*, 2017; van Kleunen *et al.*, 2015). However, as mentioned above, these datasets rarely distinguish those alien species which are invasive (Richardson *et al.*, 2010), and, as such, in this section the term alien species is used. Island and coastal mainland regions have higher alien species richness (i.e., total number of species) than mainland regions (Dawson *et al.*, 2017; **Figure 1.4**). Alien species richness is dependent on the number of different species introduced to a given location, often referred to as colonization pressure (Blackburn *et al.*, 2020; Lockwood *et al.*, 2009; **Glossary**).

Not all alien species transported beyond their natural ranges establish sustaining populations (Cassey *et al.*, 2018; Richardson *et al.*, 2010). Propagule pressure (**Glossary**) is a measure of introduction intensity comprising both the number of individuals introduced per introduction event (propagule size) and the frequency of introduction events (Cassey *et al.*, 2018; Colautti *et al.*, 2006; Lockwood *et al.*, 2005). Given suitable environmental conditions, the total number of individuals of a particular alien species that are introduced has been shown to be positively correlated with the establishment success of alien populations (Colautti *et al.*, 2006; Lockwood *et al.*, 2009). The more individuals released, the greater probability that the population will have sufficient genetic variation to adapt to local conditions and establish self-sustaining populations (Blackburn *et al.*, 2009).

Social and economic factors, including gross domestic product per capita and population density (**Chapter 3**, **sections 3.2.2** and **3.2.3**), are important in determining alien species richness globally (Dawson *et al.*, 2017). High trade and transport connectivity amongst regions which have similar environmental conditions can also be important in predicting the risk of invasive alien species (**Glossary**; Capinha *et al.*, 2014; Cope *et al.*, 2019; Early & Sax, 2014; Fitzpatrick *et al.*, 2007; Li *et al.*, 2014; Parravicini *et al.*, 2015) and describing global patterns of alien species richness (**Chapters 2** and **3**).

1.3.3 Drivers of change in nature affecting invasive alien species

Direct and indirect drivers of change refer to all external factors that affect nature and consequently nature's

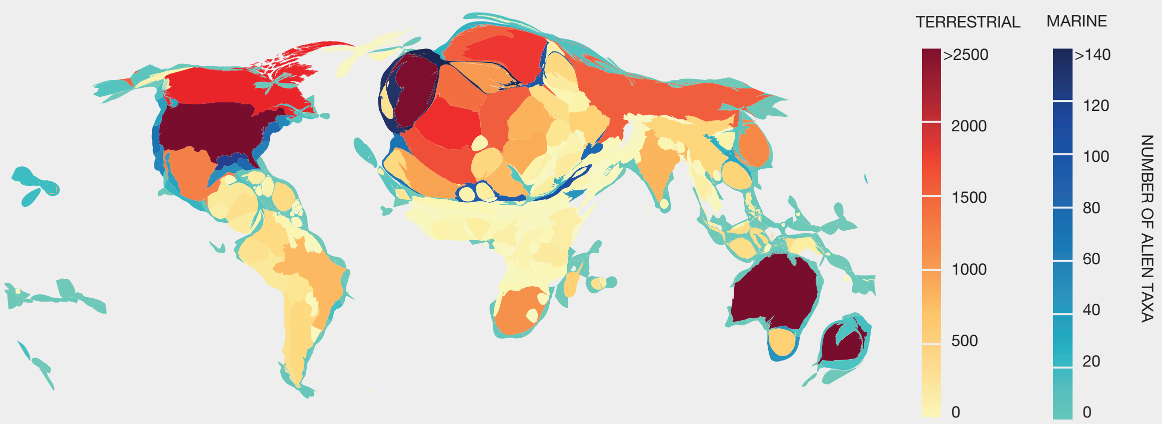


Figure 1 4 **Number of documented alien taxa in terrestrial (including freshwater) and marine environments.**

The size of regions is scaled proportionately by the number of documented alien taxa. Gaps for global alien species records are documented in **Chapter 2**. Data source: Seebens (2021).

contributions to people and good quality of life (Brondizio *et al.*, 2019). Direct drivers may be both human (anthropogenic) and non-human factors. Direct drivers affect nature directly in physical ways and include land or sea-use change, direct exploitation of natural resources, climate change, pollution, and invasive alien species. Indirect drivers are human actions that act on and alter direct drivers and other indirect drivers. Indirect drivers do not physically affect nature or nature's contributions to people, but they are the underlying cause of direct anthropogenic drivers. Indirect drivers include the role of institutions and governance (**Glossary**) systems, economic policies, and demographic, technological, and cultural influences.

The categories of indirect and direct drivers used throughout the IPBES invasive alien species assessment are based on the IPBES conceptual framework (Díaz *et al.*, 2015) with modifications specifically relevant to biological invasions and invasive alien species outlined in **Chapter 3**, **section 3.1.2**. The importance of interactions between invasive alien species and other drivers of change is acknowledged across the IPBES assessments (IPBES, 2018d, 2018e, 2018f, 2018g, 2019), and MacDougall & Turkington (2005) note that some invasive alien species may be considered passengers of global change because they only persist in an ecosystem through continued human disturbance. However, it is also important to recognize that alien species are themselves a component of biodiversity; they may be affected by other direct and indirect drivers while also interacting with native biodiversity and other alien species (**Chapter 3**, **section 3.3.5**).

Drivers may act alone or interact with each other to varying degrees, leading to additive or multiplicative effects (**Chapter 3**; Díaz *et al.*, 2018; Newbold *et al.*, 2015; Sala *et al.*, 2000) in which it is difficult to determine the relative importance of one driver over another (**Boxes 1.3** and **1.4**). For example, land-use changes are widely recognized as playing a role in promoting invasive alien species (IPBES, 2018c; Mooney & Hobbs, 2000). However, the role of indirect and direct drivers, and the complex interplay amongst them, will vary through the stages of the biological invasion process (**section 1.4**; **Glossary**). This complexity is rarely addressed within studies on invasive alien species but is increasingly recognized as an important consideration in understanding biological invasions and deriving solutions to mitigate or manage invasive alien species. It is important to recognize that drivers of change in nature such as land- and sea-use change, climate change and invasive alien species act at different temporal and spatial scales (**Chapter 3**; also **Figure 1.9** in **section 1.5**; Bonebrake *et al.*, 2019).

1.3.4 What are impacts in the context of invasive alien species?

For the purposes of this assessment, an impact is defined as a measurable change to nature, nature's contributions to people, and/or good quality of life (**Figure 1.1**; Ricciardi *et al.*, 2013; **Chapter 4**, **section 4.1.2**). It is useful to discriminate between measurable changes in physical or social parameters and value-laden decisions on whether such changes are beneficial or detrimental to humans or native species (Vimercati *et al.*, 2020). Invasive alien species can cause changes in physical, chemical, and/or biological properties, which can result in an increase or decrease in a parameter or an index. Such change may be considered as a harmful impact with respect to nature if whole ecosystems and communities are affected, or if other species are negatively (e.g., reduction in their performance and/or population size, or extinction) or positively (e.g., increase in their performance and/or population size, or establishment of new populations) affected. Impacts can also be considered as harmful (negative) or beneficial (positive) for humans if people suffer or gain from changes in nature's contributions to people or constituents of good quality of life

Box 1 3 Interactions between invasive alien species and climate change as drivers of biodiversity loss.

The IPBES-IPCC Co-Sponsored Workshop Report on Biodiversity and Climate Change (Pörtner *et al.*, 2021) recognized that climate change and biodiversity loss are interconnected and share common drivers through human activities. Although the outcomes of interactions between climate change and invasive alien species on community level processes is poorly understood (Robinson *et al.*, 2020), disproportionate changes in community composition across trophic levels are predicted to decrease species diversity and stability (Zarnetske *et al.*, 2012). As an example, climate change is anticipated to affect top predators more strongly than lower trophic levels, leading to an increase in herbivores and a decrease in plants (Zarnetske *et al.*, 2012). It is evident that the ongoing unprecedented changes in climate will alter the interactions between native and alien species (**section 1.6.8**). Interactions amongst drivers of change in nature, including climate change and invasive alien species but also other drivers, can generate complex feedback loops (Sinclair *et al.*, 2020; **Glossary**) with pronounced and unpredictable outcomes on evolutionary and ecosystem level processes (Pörtner *et al.*, 2021; **Chapter 3**, **section 3.5**).

Box 1 4 **Climate change, fire, and invasive alien plants.**

Many regions are experiencing unprecedented fire regimes because of human-driven ignition, coupled with intense droughts and record high temperatures associated with human-induced climate change (Bowman *et al.*, 2020; Kelly *et al.*, 2020). Undoubtedly, the increase in frequency and intensity of fires is threatening ecosystems and good quality of life in almost all parts of the world (Bowman *et al.*, 2011; **Figure 1.5**). Invasive alien species can worsen the situation by adding fire-prone fuel, which can increase not only the fuel quantity but also its flammability and its spatial continuity (Brooks *et al.*, 2004; Gaertner *et al.*, 2014). Studies have found that in several biomes, including tropical, temperate and Mediterranean regions, invasive alien plants may benefit from fires but can also act as promoters of more intense and frequent fire regimes, potentially causing more carbon release into the atmosphere (Nuñez *et al.*, 2021). In the Cerrado forest of Brazil, for example, *Melinis minutiflora* (molasses grass) and *Urochloa brizantha* (palisade grass) introduced in the 1800s are more prone to fire and although fire is a natural disturbance of this ecosystem, invasive alien grasses increase the frequency and intensity of fires (Damasceno & Fidelis, 2020). In Mediterranean climates and other semi-arid and arid ecosystems, some land-use practices, such as overgrazing, have resulted in significant increases in invasive alien European grasses such as *Bromus tectorum* (downy brome) that increase fuel load, continuity, and flammability. These conditions create a positive feedback loop between severe fires and the invasion of *Bromus tectorum* that results in multiple negative changes of natural grasslands and shrub steppe ecosystems and services (e.g., Western North America; see Pyke *et al.*, 2016). In areas with Mediterranean and temperate climates, especially in the southern hemisphere, shrubs and trees native to fire-prone ecosystems may cause extreme changes in fire regimes. In southern Africa and southern South America, Australian species of *Acacia* have shown to spread rapidly after fires and their biomass can fuel more intense fires (Le Maitre *et al.*, 2011). Similar positive feedback loops between invasive alien species and fires have been observed for *Pinus* across several ecosystems in the southern hemisphere (Cóbar-Carranza *et al.*, 2014; Franzese & Raffaele, 2017; Taylor *et al.*, 2017). Fire-prone invasive alien plants are likely to continue to spread under the more extreme climate scenarios and with the anticipated increase in conditions favourable to fire (Hurteau *et al.*, 2014). Consequently, these invasive alien plants are predicted to play a role in promoting more intense fire regimes with potential impacts on carbon cycling and further potential synergies with climate change.

Figure 1 5 **Invasive alien plants increase fire intensity and spread.**

A volunteer in Chile is trying to control a wildfire in an area invaded by *Genista monspessulana* (Montpellier or French broom). Photo credit: Guillermo Salgado Sánchez – CC BY 4.0.

(**Chapter 4**, **sections 4.1.3**, **4.4** and **4.5**; García-Llorente *et al.*, 2008; Pyšek & Richardson, 2010; F. Williams *et al.*, 2010). It is important to acknowledge that the outcomes of assessments of the benefits or positive impacts of invasive alien species should not be used to balance or offset the harmful or negative impacts, which may be irreversible including ecosystem transformation (Lockwood *et al.*, 2023; **Chapter 4**). Invasive alien species can have direct or indirect adverse impacts in their new environment even if their populations are not established or conversely can have negligible impacts even when established and widespread (**Glossary**; Jeschke *et al.*, 2013). While most literature on invasive alien species refers to the detrimental effects on ecological processes in terrestrial, freshwater and marine environments, new evidence is revealing the devastating effects on social (Bacher *et al.*, 2018; Gallardo *et al.*, 2019) and economic aspects (Diagne *et al.*, 2020). There is consensus among the scientific community that impacts of invasive alien species cannot be understood independently of other drivers of change in nature and that ecological, social, and economic aspects are also closely intertwined (Pyšek, Hulme, *et al.*, 2020; Shackleton, Shackleton, *et al.*, 2019).

Previous IPBES assessments have concluded that increased biotic homogenization (**Glossary**), or loss of biotic uniqueness, of biological communities is a major negative impact of invasive alien species which can result in the introduction and establishment of further alien species (IPBES, 2018d, 2018e, 2018f, 2018g, 2019). Local community assemblages are becoming more similar to each other on average, and this biotic homogenization (Finderup Nielsen *et al.*, 2019; McKinney & Lockwood, 1999; Yang *et al.*, 2021) has also been referred to as the "anthropogenic blender" (Olden, 2006). A recent review highlighted a consistent trend of decreasing taxonomic and phylogenetic diversity globally, providing strong evidence of widespread biotic homogenization (D. Li *et al.*, 2020). The consequences of biotic homogenization for ecosystem processes and nature's contributions to people can be substantial, but are often context specific, are hard to predict, and remain understudied. Ongoing environmental transformation is reducing the ability of ecosystems to withstand disturbance, including the arrival of invasive alien species, and so leading to decline in the resilience (**Glossary**) of natural systems (Dasgupta, 2021).

The introduction of one invasive alien species can facilitate invasion by another (**Chapter 3**, **section 3.3.5**; Simberloff & Von Holle, 1999). In some cases, this has led to an increasing rate of establishment and consequently communities of interacting invasive alien species are becoming increasingly common (Jackson, 2015; Simberloff & Von Holle, 1999). This facilitation is more likely to occur when a high number of species are introduced to an area (e.g., islands) or for alien species that are already known to interact with one another (e.g., species that co-occur within the native range or previously invaded ranges), such as pests and parasites. Indeed, parasites and pathogens are frequently introduced into new communities alongside invasive alien species and are implicated in altering the outcome of biological invasions by changing the strength of interactions between alien and native species (Dunn & Hatcher, 2015; **Box 1.14** in **section 1.6.7.2**). Co-occurring and interacting invasive alien species may amplify and exacerbate negative impacts. Indeed, biotic facilitation (**Glossary**), the synergistic interactions amongst different alien species within an invaded ecosystem, can lead to extreme adverse effects on ecosystem functions, which have been termed "invasional meltdown" (Simberloff, 2006; Simberloff & Von Holle, 1999; **Glossary**). However, in some cases interactions amongst invasive alien species can mitigate the adverse effects, for example when a predator is introduced and reduces the population of the prey of the invasive alien species (**Chapter 3**, **section 3.3.5**; Braga *et al.*, 2018; Facon *et al.*, 2006; Jackson, 2015).

The effects of an invasive alien species on an invaded biotic community will increase as the density of the invading organisms increases (Shea & Chesson, 2002). Effects on and responses of the resident species will in turn determine whether the community provides opportunities for invasive alien species (Parker *et al.*, 1999). However, while it is recognized that the outcome of biological invasions can be partially explained by the traits of alien species (invasiveness, i.e., the intrinsic biological characteristics of the species that result in the ability to invade a particular ecosystem) and characteristics of the recipient community (invasibility, i.e., susceptibility of an ecosystem to be invaded by one or multiple species), high levels of uncertainty (Leung *et al.*, 2012) are often a feature of predictions on the dynamics of invasive alien species (Facon *et al.*, 2006; Hui & Richardson, 2019). It is critical to integrate characteristics of the invading species alongside characteristics of the recipient habitats to account for the context within which the biological invasion is occurring (Foxcroft *et al.*, 2011).

Invasive alien species may reduce the phylogenetic distance among species within a community and, although in some cases they may increase the phylogenetic diversity within local sites, they can reduce phylogenetic diversity between sites (D. Li *et al.*, 2020). Ecosystem function is influenced by phylogenetic diversity (Cadotte *et al.*, 2012); ecosystems comprising community assemblages with higher phylogenetic diversity are considered to be more resilient to disturbance because they have the evolutionary potential to adapt to changing environmental conditions (D. Li *et al.*, 2020). The diversity and relative abundances (evenness) of species may strongly affect ecosystem function for community assemblages comprising combinations of functionally different species with low niche overlap (Cadotte *et al.*, 2012). While it is difficult to

Box 1 5 Role of invasive alien species within novel or emerging ecosystems.

Changes in the composition of communities as a consequence of invasive alien species will lead the emergence of new species combinations. Ecosystems containing these new species combinations are termed "novel ecosystems" or "emerging ecosystems" (Hobbs *et al.*, 2006). A broad range of examples document the emergence of novel ecosystems specifically in the context of biological invasions leading to new species combinations (Haram *et al.*, 2021; Lindenmayer *et al.*, 2008; Lugo, 2004; Mascaro *et al.*, 2008; Wilkinson, 2004). The adverse consequences of these changes include hybridization (e.g., between *Sporobolus maritimus* (small cordgrass) and *Sporobolus alterniflorus* (smooth cordgrass) leading to the emergence of the invasive alien *Sporobolus anglicus* (common cordgrass)), species declines (e.g., brown tree snake decimation of the forest bird species in Guam; Rodda & Savidge, 2007), or ecosystem-level change (e.g., changes in nutrient cycles, fire cycles or hydrology; Ehrenfeld, 2010; Ramakrishnan & Vitousek, 1989; Simberloff, 2011; Vilà *et al.*, 2011; Vitousek, 1986). However, novel ecosystems have shown to be beneficial in some contexts (Munishi & Ngondya, 2022) including, for example, by restoring ecosystem processes (Ewel & Putz, 2004; Lugo, 2004; C. E. Williams, 1997) or by providing nature-based solutions to mitigate environmental change (Munishi & Ngondya, 2022) although it is recognized that more evidence is needed for the latter (Turner *et al.*, 2022). Furthermore, context-specific adaptive governance (**Glossary**; **Chapter 6**, **Table 6.6**) coupled with pathway management (**Glossary**; **Chapter 5**, **section 5.4.3.1**) and understanding of drivers (**Chapter 3**) and more broadly the biology of alien species, including their interactions with native species, is considered critical to success of nature-based solutions for managing biological invasions (Munishi & Ngondya, 2022).

The formation of novel ecosystems that include invasive alien species has led to discussions about the implications of resulting compositional and ecological changes (e.g., Hobbs *et al.*, 2014; Murcia *et al.*, 2014). Perceptions (**section 1.5.2**) depend on many factors including concerns over environmental and societal impacts but also differing cultural values toward "nativeness" and "exoticism" and how such beliefs develop over time (Higgs, 2003). The range of perceptions may also be based on how effective the actions are likely to be in reversing the changes caused by invasive alien species. On one side of the spectrum, reversal of the novel state generated by alien species is viewed as a useful, morally necessary, and achievable goal (Hallett *et al.*, 2013; Hobbs *et al.*, 2006). On the other side of the same spectrum, the transition to a novel system due to alien species impacts is viewed as irreversible when a system has crossed an ecosystem restoration (**Glossary**) threshold (Hallett *et al.*, 2013; Hobbs *et al.*, 2006). The latter is the case for most marine biological invasions, where post-establishment management actions are mostly unsuccessful and invasive alien species can alter ecosystem functions and ultimately transform the entire landscape (E. Sala *et al.*, 2011). As an example, the snail *Littorina littorea* (common periwinkle), first recorded in the mid-1800s in the north-west Atlantic subsequently spread throughout the Atlantic coast of North America, altering the diversity, abundance and distribution of many benthic species on rocky and soft shores (Carlton, 1992).

Irreversible impacts are also likely to occur in scenarios where invasive alien species remain undetected for long periods of time. These historical biological invasions hamper our ability to recognize pre-existing native landscapes and ecosystems causing what is called "ecological mirages" (Bortolus *et al.*, 2015). The historical introduction of *Sporobolus alterniflorus* to the east coast of South America during the 1800s modified the pre-existing and extensive bare mudflats into vegetated salt marsh areas, leading to shifts in bird, fish and invertebrate biodiversity, with concomitant trophic cascades, but these changes were long overlooked (Bortolus *et al.*, 2015).

Acknowledging the uncertainty of outcomes of novel ecosystems and the potential for invasional meltdown (**Chapter 3**, **sections 3.1.3.2** and **3.3.5**), it is desirable to adopt a cautious and context-specific approach when considering the impacts of alien species and of the novel ecosystems they generate (Hobbs *et al.*, 2006), including the potential role of novel ecosystems as nature-based solutions to mitigating other drivers of change in nature (Seddon *et al.*, 2021). This uncertainty also highlights the value of pragmatism when recommending management strategies, and the benefits of engaging all stakeholders with available evidence to consider desirability, cost, and resource availability (**Chapters 5** and **6**; Hallett *et al.*, 2013; Miller & Bestelmeyer, 2016). There are many ways in which alien species interact with one another and with native species (Hui *et al.*, 2021). Novel mutualistic interactions (pollination, seed dispersal and plant-microbial symbioses) amongst alien species have been shown to facilitate other invasive alien species(Traveset & Richardson, 2014) leading to cascading effects that alter ecosystem functioning **(Box 1.11)**. Less attention has been given to interactions between alien and native species which lead to benefits, or indeed reductions in the magnitude of adverse impacts of interacting alien species (Liu *et al.*, 2018; Ross *et al.*, 2004), but it is acknowledged that beneficial interactions are also important in determining the outcomes of biological invasions on communities and consequently ecosystem function (Braga *et al.*, 2018; Halpern *et al.*, 2007; Viana *et al.*, 2019). The outcomes of species interactions are highly context-dependent (Lord *et al.*, 2017) and other drivers of change in nature will alter the population dynamics of alien and native species with consequences for eco-evolutionary and community-level processes which can be difficult to predict (Facon *et al.*, 2006; Robinson *et al.*, 2020).

quantify niche overlap and functional differences among multiple species, phylogenetic diversity can be used as a proxy of similarities and differences amongst species (Cavender-Bares *et al.*, 2009). Species-specific traits or human-mediated processes have been shown to be more important sources of variation in establishment and spread of invasive alien species than phylogenetic diversity (**Chapter 3**; Diez *et al.*, 2008). However, it is important to include multiple facets of biodiversity when assessing impacts, and phylogenetic diversity can be used as metric for predicting multifunctionality of ecosystems (Lishawa *et al.*, 2019). Innovative approaches integrating species distributions, traits, phylogenies, and interaction networks incorporating feedback loops will contribute to better understanding of biodiversity change (Pollock *et al.*, 2020) including predicting the outcomes of biological invasions (Hui & Richardson, 2019).

Since invasive alien species interact with resident species in evolving ecosystems **(Box 1.5)**, elucidating the complex adaptive networks these invasive alien and resident species form is critical to underpin understanding of the dynamics of invasive alien species and management of biological invasions. Network ecology embraces the multitude of biotic interactions within a framework of feedback loops which affect species persistence and coexistence (Borrett *et al.*, 2014; Hui *et al.*, 2016) and ultimately the functioning of ecosystems (Harvey *et al.*, 2017). Emerging insights in understanding the influence of human decisions, perceptions and management efforts within the context of ecological networks will improve forecasts on the response of networks to invasive alien species (Kueffer, 2017).

Ecological impacts of invasive alien species include adverse effects on biodiversity and also on nature's contributions to people (**Chapter 4**, **sections 4.3** and **4.4**). Invasive alien species can lead to extreme disruptions in the good quality of life of local communities (**Chapter 4**, **section 4.5**) either by indirect impacts on human health (e.g., introduced mosquitoes and disease; see **Box 1.14** in **section 1.6.7.2**), reduction of food security (e.g., invasive alien species as weeds in crop systems) or through degradation of habitats on which people depend (e.g., fire regime shifts caused by some invasive alien plants that are particularly flammable). As with any ecosystem change, there are cases where invasive alien species may provide opportunities for people to adapt and take advantage of the new conditions the species can provide. Production of firewood, new food sources and strengthening of aesthetic and cultural values have been recognized as beneficial outcomes of biological invasions (Shackleton, Shackleton, *et al.*, 2019). However, the overall impact on nature's contributions to people and good quality of life is hard to assess, as these species may have also disrupted the traditional and cultural ways of living of many Indigenous Peoples and local communities (**Chapter 4**, **section 4.6**).

1.4 BIOLOGICAL INVASION PROCESS

Over the past thirty years, different approaches to describe biological invasions have been developed (Colautti & MacIsaac, 2004; Leung *et al.*, 2012; Rejmanek & Richardson, 1996; Rojas-Sandoval & Acevedo-Rodríguez, 2015; Williamson, 1996; Williamson & Fitter, 1996). The unified framework for biological invasions **(Figure 1.6)** emerged from the integration of key features from across these commonly used frameworks and represents a single conceptual model that can be applied to all human-mediated biological invasions (Blackburn, Pyšek, *et al.*, 2011). This framework is used throughout the IPBES invasive alien species assessment.

The unified framework divides the biological invasion process into a series of stages (transport, introduction, establishment, and spread), recognizing the need for a species to overcome the barriers (geography, captivity or cultivation, survival, reproduction, dispersal, and environmental) that obstruct transition between each stage. Different factors may be advantageous in allowing species to pass through each stage **(Figure 1.6)**. The two barriers, survival and reproduction, recognize that the establishment stage is a population process, and establishment of a viable population requires self-sustaining populations encompassing multiple generations. **Chapter 4** provides a synthesis of the environmental, economic and social impacts which can occur throughout the biological invasion process. Evolutionary processes and mechanisms, including evolutionary history, founder effects, and hybridization, are also relevant (Dlugosch *et al.*, 2015; Estoup *et al.*, 2016; Facon *et al.*, 2006; Hufbauer *et al.*, 2012; Zenni *et al.*, 2017) and considered further within **Chapter 2**, **Box 2.3**.

1.4.1 Transport

Transport is the first stage in the biological invasion process (Williamson, 1996). Species have native geographic distributions with limits imposed by natural constraints, both biotic and abiotic. Human activities, such as shipping for trade, agricultural practices, and ornamental planting, can result in the movement of species beyond the barrier(s) that define these natural limits (**Chapter 3**). Humans can deliberately or inadvertently break down the natural barrier(s) which otherwise define these natural limits in the global distribution of species. This barrier is termed "geography" (Rojas-Sandoval & Acevedo-Rodríguez, 2015) in the unified framework as it is typically a physical feature (e.g., a mountain range or ocean) or a climatic barrier through which a species cannot normally disperse. However, the barrier may also be biogeographical, if distributional limits are imposed by biotic factors such as the presence

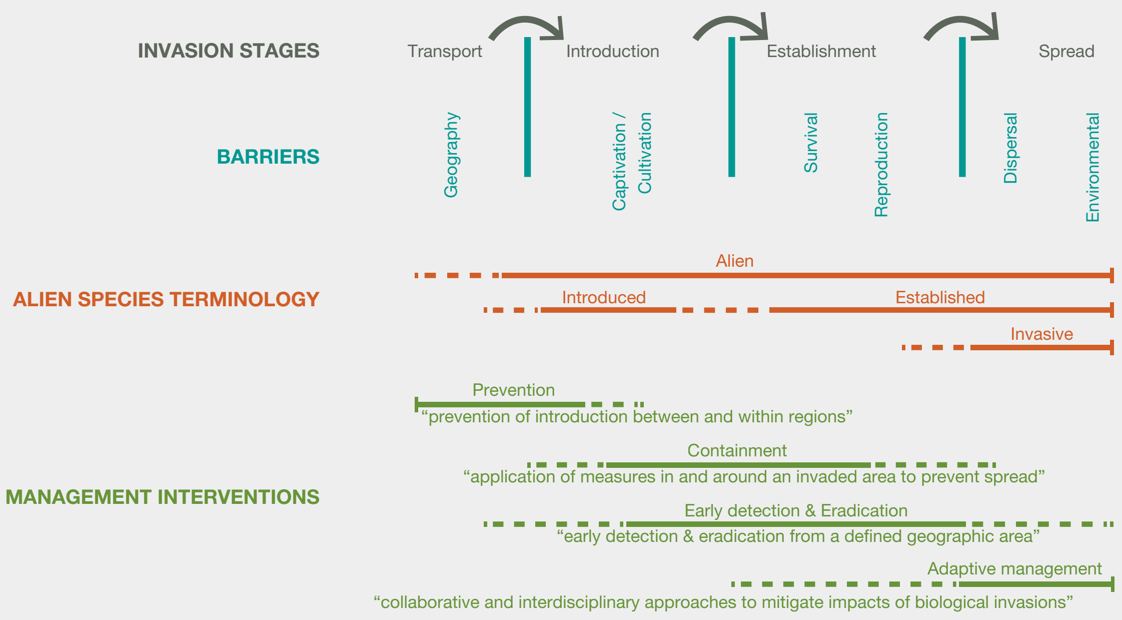


Figure 1 6 **The biological invasion process.**

Alien species terminology adopted throughout the assessment. The status of the species is reflected in the change in terminology and depends on the stage within the biological invasion process. Each stage is separated from the next by a barrier which a species must overcome in order to pass to the next stage. Alien species at different stages are associated with different management interventions. Solid horizontal lines represent the core application of the terminology, or management interventions and broken lines indicate extended application of the terminology in some contexts and where suitable management interventions may have some relevance. Adapted from Blackburn, Pyšek, *et al.* (2011), https://doi.org/10.1016/j.tree.2011.03.023, under Copyright 2011 Elsevier Ltd.

Box 1 6 Pathways of introduction of invasive alien species.

Pathways describe the many ways in which an alien species can be intentionally or unintentionally introduced through human activities from one geographical location to another (Hulme *et al.*, 2008; Pyšek *et al.*, 2011). Recognizing the importance of linking pathways to management or legislative options, a pathway scheme was developed by Hulme *et al.* (2008) that coupled policy options with the broad mechanisms by which alien species could be introduced to a region. The Conference of the Parties to the CBD subsequently adopted (and refined) the pathway scheme proposed by Hulme and colleagues (Hulme, 2014; Hulme *et al.*, 2008) to give a unified system for categorizing alien species pathways (CBD, 2014). The CBD Pathway Scheme distinguishes intentional and unintentional introductions, the six broad mechanisms of introduction (categories) and a number of corresponding subcategories. Furthermore, Saul *et al.* (2017) have published guidance for interpretation of the categories in introduction pathways, including for the six broad mechanisms of introduction:

Release in nature: intentional introduction of alien species for the purpose of human use in the natural environment;

Escape: unintentional movement of alien species from confinement (e.g., in zoos; aquaria; botanic gardens; agriculture; horticulture; aquaculture and mariculture facilities; scientific research or breeding programmes; or from keeping as pets) into the natural environment;

Transport-contaminant: unintentional movement of alien species as contaminants of a commodity that is intentionally transferred through international trade, development assistance, or emergency relief;

Transport-stowaway: unintentional movement of alien species attached to transporting vessels and associated equipment and media;

Corridor: unintentional movement of alien species into a new region following the construction of transport infrastructures in whose absence spread would not have been possible;

Unaided: secondary natural dispersal (**section 1.4.4**) of alien species that have been introduced by means of any of the foregoing pathways.

of competitors, predators, parasites and pathogens, or the absence of mutualists. Barriers to dispersal promote diversification by driving important evolutionary processes (e.g., speciation) and as such environmental conditions that prevent organisms from dispersing have far-reaching consequences for the organization of life on earth (Caplat *et al.*, 2016). The ways in which alien species are intentionally or unintentionally introduced through human activities from one geographical location to another are termed "pathways" (Hulme *et al.*, 2008; Pyšek *et al.*, 2011). An alien species may arrive within a new region through the importation of a commodity, arrival of a transport vector (physical means or agent, such as ship, train, aircraft, or other vehicle), which an alien species moves in or on (IUCN, 2017), and/or natural spread from a previously invaded region (Hulme *et al.*, 2008). These three mechanisms of arrival can be subdivided into six major pathways **(Box 1.6)**. It is evident that the pathways through which alien species are transported and introduced to new regions are changing over time (Essl *et al.*, 2015; Hulme *et al.*, 2008) and it is apparent that some of the most problematic invasive alien species arrive through multiple pathways (Essl *et al.*, 2015; Saul *et al.*, 2017) and repeated introductions (J. R. U. Wilson *et al.*, 2009). The movement of alien species may be facilitated by a broad range of human factors, or drivers of change, especially those related to the economy, human demography, and land-use (**Chapter 3**).

1.4.2 Introduction

A species may be moved to a location beyond its natural distributional limits but will only go on to invade an area if it is introduced beyond captivity and cultivation from that location (Williamson, 1996). To become introduced, individuals of that species must overcome the (sometimes literal) barriers imposed by captivity or cultivation (Rojas-Sandoval & Acevedo-Rodríguez, 2015). A deliberate (intentional introduction) act may be with the aim of establishing an alien species, for example if the species can be considered economically (e.g., game species) or environmentally (e.g., biological control agents, **Glossary**) or culturally (e.g., landscape gardening; van Kleunen *et al.*, 2018) beneficial. Over time, a wider understanding of the harm that invasive alien species can cause (Pyšek, Hulme, *et al.*, 2020) led to the conclusion that most introductions are not deliberate, but are unintentional. Important anthropogenic factors, or drivers, that may facilitate the introduction of invasive alien species include escape from captivity (e.g., pet animal escapes, seed spread from botanical gardens, larvae or adults that escape from aquaculture facilities) or escape by stowaways (e.g., organisms in ballast water), although some can result from intentional liberation of individuals into a novel environment (e.g., ceremonial release of animals) (Dyer *et al.*, 2017; Magellan, 2019; Pyšek, Hulme, *et al.*, 2020; Simberloff *et al.*, 2013; **Chapter 3**).

1.4.3 Establishment

Introduced species will fail to become invasive if they are unable to produce a self-sustaining and viable population in the new location, a process that is termed "establishment" (Williamson, 1996). This stage in the biological invasion process requires that introduced individuals both survive and reproduce in the new environment, and hence that barriers to survival and reproduction are overcome (Pyšek, Bacher, *et al.*, 2020; Rojas-Sandoval & Acevedo-Rodríguez, 2015). Therefore, as mentioned in **section 1.3.2**, biological invasions are a function of propagule pressure, colonization pressure, abiotic characteristics of the invaded ecosystem and biotic characteristics of the recipient community and invading species (Catford *et al.*, 2009; Lockwood *et al.*, 2009) including ecological and evolutionary change (Facon *et al.*, 2006).

The number of individuals introduced into a new environment has been the most consistently described and widespread correlate of establishment success of alien species (Blackburn, Prowse, *et al.*, 2011; Lockwood *et al.*, 2005). Indeed, propagule pressure is considered the most reliable predictor of biological invasion success (Colautti *et al.*, 2006). As already described, the term propagule pressure incorporates both the number of individuals released in one introduction event and the number of such events (Lockwood *et al.*, 2005). Small introduced populations, with a few notable exceptions (Briski *et al.*, 2018; Roman & Darling, 2007), are likely to fail to establish because of constraints of demography, genetics or environmental variation, even if the location is suitable for their survival and reproduction (as is also the case for small populations of threatened native species) (Cassey *et al.*, 2018; Duncan *et al.*, 2014; Lockwood *et al.*, 2005).

The outcome of a specific introduction and establishment is dependent on resource availability, interactions with other species including natural enemies (predators and parasites), and the abiotic environmental conditions (Catford *et al.*, 2009; Roy & Lawson Handley, 2012; Shea & Chesson, 2002). These factors all vary in time and space and can be modified by human influences or drivers of change in nature (**Chapter 3**) and natural disturbances (Catford *et al.*, 2012). The relative importance of these factors varies between species. As an alien species increases in population density, it will influence the invaded locality through interactions with other species within the community. The process of biological invasion is dynamic and specific outcomes of interactions vary over time and with context including the responses of humans to the invasive alien species, which can range from adaptation to management including eradication and ecosystem restoration **(Box 1.7)**.

The concept of invasibility, the susceptibility of a community to become invaded by one or several species, has been described as an intrinsic community and ecosystem attribute, but this view has been challenged because the lack of available information on species that have failed to establish makes it difficult to infer whether some species are more invasive or some habitats more invasible than others (Colautti *et al.*, 2006; Zenni & Nuñez, 2013). Furthermore, invasiveness of an alien species and the invasibility of the recipient ecological network are interlinked (Hui *et al.*, 2021); establishment success is a function of the interaction between traits or invasiveness of the species (e.g., behaviour, physiology, life history) and invasibility of the environment (e.g., climate, habitat) (Abramides *et al.*, 2011),

Box 1 7 Ecosystem restoration enhancing resilience to invasive alien species.

Ecosystem restoration is defined as any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state (IPBES,[3] e.g., **Figure 1.7**) – i.e., assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed – and is often used to reinstate ecosystems that have been altered by invasive alien species. An exciting extra role for ecosystem restoration is to prevent the establishment and spread of invasive alien species in the first place. Indeed, there is increasing interest in using restoration to enhance ecosystem resilience to perturbations as environmental change accelerates.

Invasive alien species are recognized as one of five major drivers of change in nature, with adverse impacts on nature and also, in some cases, nature's contributions to people and good quality of life (**Chapter 4**). As such, management of biological invasions is critical to achieving ecosystem restoration (**Chapter 5**, **section 5.5.7**). However, there is also considerable evidence of invasive alien species as "passengers" of change (S. D. Wilson & Pinno, 2013). Restoring ecosystems to prevent the establishment and spread of invasive alien species is most obviously beneficial under the so-called "Passenger Model", under which invasive alien species are facilitated by anthropogenic environmental change – such as disturbance or eutrophication (**Chapter 3**, **sections 3.3.1** and **3.3.3**). In this case invasive alien species are "passengers" that benefit from the altered environment rather than themselves driving change (MacDougall & Turkington, 2005).

Invasive alien species are frequently a problem during ecosystem restoration, and much research focuses on how to control them. By contrast, studies of the ability of restored ecosystems to prevent the establishment and spread of invasive alien species are few, and most assess resistance during the early stages of ecosystem restoration. For example, Foster *et al.* (2015) found that following experimental additions of invasive alien species, including the highly invasive alien legume *Lespedeza cuneata* (sericea lespedeza), restored American prairie strongly limited invasive alien species compared to unrestored prairie. In general, a high native diversity might be expected to increase resistance to invasive alien species (Byun *et al.*, 2018). However, there is a lack of evidence about the ability of ecosystem restoration to limit biological invasions over the long-term and at large scales.

2021 marked the start of the United Nations-sponsored Decade on Ecosystem Restoration, acknowledging that ecosystem restoration could become central in efforts to resist and effectively prevent biological invasions. Ecosystem restoration has many other benefits, including the enhancement of ecosystem functions and benefits to people, the provision of habitat for native species, and resilience to ongoing environmental change.

3. IPBES glossary: https://ipbes.net/glossary

Figure 1 7 Restoring calcareous grassland in southern England.

Left: flower rich calcareous grassland following ecosystem restoration. Right: *Ochlodes venata faunus* (large skipper) after ecosystem restoration. Photo credit: Maico Weites – CC BY 4.0.

but crucially also depends on human actions (Duncan *et al.*, 2003; Redding *et al.*, 2019) and on many different and interacting drivers of change in nature (**Chapter 3**, **section 3.5**).

1.4.4 Spread

The next stage in the biological invasion process is known as spread, whereby individuals from an established population disperse across the new environment (Williamson, 1996), increasing the size of the geographic distribution of the alien species. An alien species can spread in various ways, such as through natural dispersal or transport alongside human activities (**section 1.4.1**). Spread requires the alien species to overcome a barrier imposed by limits to dispersal (e.g., the distance between suitable habitat patches), and a barrier imposed by environmental suitability (Rojas-Sandoval & Acevedo-Rodríguez, 2015), which will tend to increase with distance from its location of establishment (Lomolino *et al.*, 2010). Spread of an alien species is a sequence of population establishments, and so environmental suitability can be viewed as presenting barriers to survival and reproduction that must be overcome in each newly colonized location. Human factors, especially those related to disturbance and the creation of corridors, may act as drivers facilitating the spread of alien species within and beyond their non-native range (**Chapter 3**, **sections 3.3.1**, **3.4.2**). It is important to note that there are often time lags, sometimes of decades or more, between introduction, establishment, and spread (Essl *et al.*, 2011; Kowarik, 1995; Seebens *et al.*, 2017).

Introduced populations of alien species can also be a source of new introductions; this is referred to as secondary spread (Bertelsmeier & Keller, 2018). Patterns of spread of alien species have been widely documented (Ascunce *et al.*, 2011; Chapman *et al.*, 2020; Keller *et al.*, 2012; Lombaert *et al.*, 2010) and the mechanisms underpinning secondary spread have been the subject of many studies and some debate (Bertelsmeier & Keller, 2018). A single introduced population can be the source of many secondary introductions and so an alien species may spread rapidly even in the absence of further direct introductions from the native range. This has led to the hypothesis that adaptations for increased invasiveness could have occurred in introduced populations compared to native populations. The term "bridgehead population" or "bridgehead effect" has been used in reference to alien species establishing a stronghold or base population prior to further incursions to other environmentally suitable regions (Lombaert *et al.*, 2010). However, evidence for adaptive evolution within bridgehead populations of introduced alien species is lacking (Bertelsmeier & Keller, 2018) but evolution can play a role in the survival and establishment of introduced species through local adaptation to the novel conditions in the invaded range (Facon *et al.*, 2006; Hufbauer *et al.*, 2012). Introduced populations can reach higher densities than those in the native range, for example because of increased resource availability in the invaded range (Catford *et al.*, 2009). The resulting high abundance, alongside other factors including ongoing introductions from the native range, increases the probability of the alien species moving to new regions with human activities, including trade networks (Banks *et al.*, 2015), providing the necessary connectivity to facilitate the secondary spread (Chapman *et al.*, 2020).

1.4.5 The management-invasion continuum

The invasion curve (**Figure 1.8**; **Glossary**) diagrammatically presents the four stages of biological invasion over time. The curve can be contextually interpreted as number of alien species, area occupied or levels of impact over space and time. It was first developed for policymakers in Australia (Victorian Government, 2010), and is now widely used across government agencies in the United States, Canada, New Zealand and Japan and by some international organizations including the IUCN. As already stated, invasive alien species often have a lag-phase during establishment (Essl *et al.*, 2011; Kowarik, 1995; Seebens *et al.*, 2017). This is followed by a dispersal phase of variable duration during which there is often logarithmic growth, up until the point at which the invasive alien species occupies a large area and so is in the widespread phase when the biophysical or socioecological negative impacts are high and affect a large proportion of the landscape/seascape (**Chapter 4**). The invasion curve highlights the importance of preventative measures (**Figure 1.8**; **Chapter 5**, **section 5.5.2**) before an invasive alien species arrives, and retaining the ability to manage an invasive alien species in the early stages of invasion after arrival. It supports understanding and decision-making of management options along the management-invasion continuum (**Chapter 5**, **sections 5.2** and **5.3**). While the invasion curve is employed widely to understand the process of biological invasions, this assessment will also utilize the IPBES conceptual framework, which is described in **section 1.6.1**.

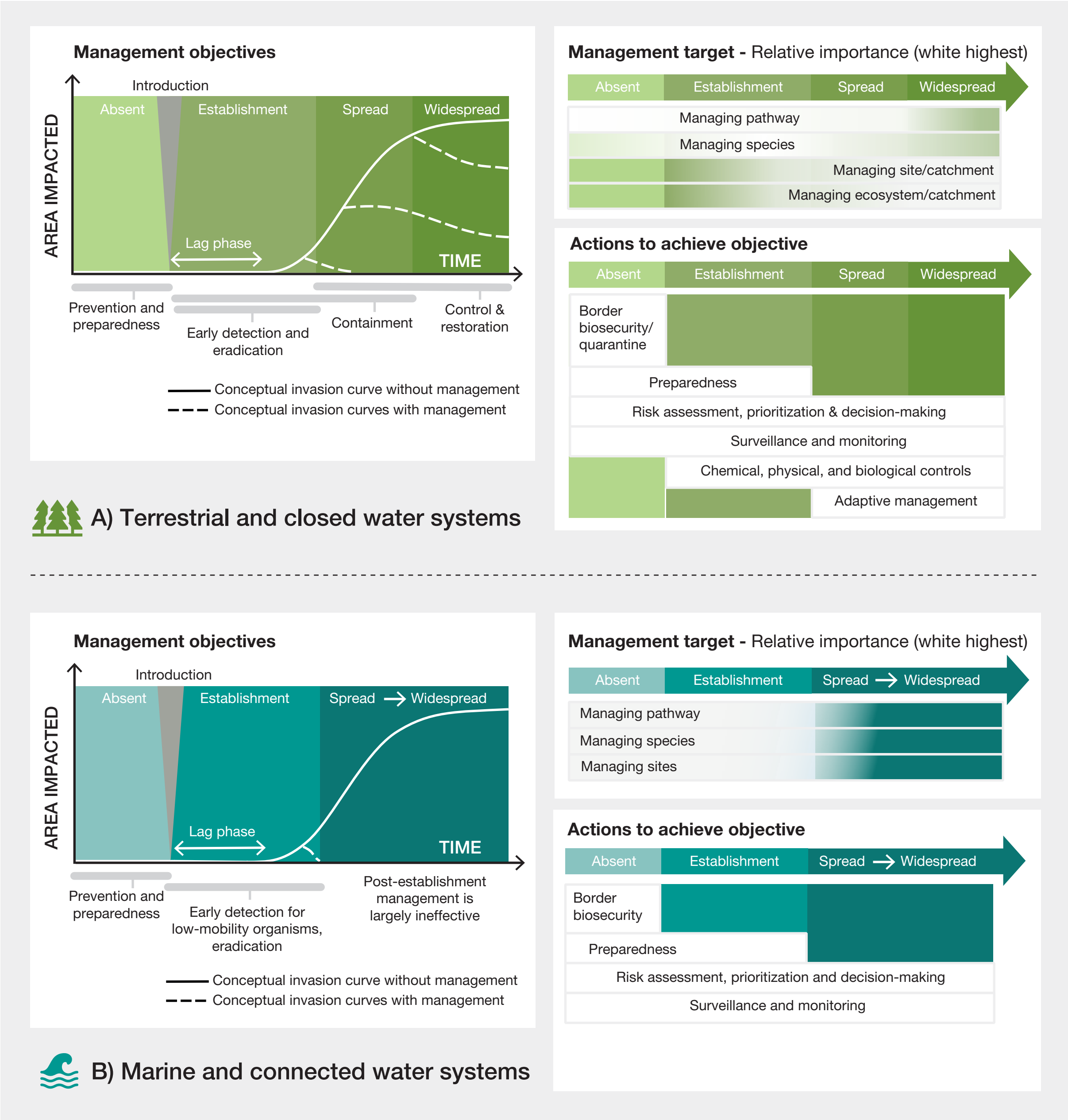


Figure 1 8 **Conceptual diagram of management-invasion continuum (see also Chapter 5, Figure 5.1).**

The generalized invasion curve without management and the expected changes in the invasion curve with appropriate management actions in **A)** terrestrial and closed water systems (including coastal systems and salt marshes) and in **B)** marine and connected water systems. White boxes indicate the optimal management objectives at each stage of the biological invasion process. Colour gradient of managing pathway, species, site and ecosystem boxes show how the relative focus generally changes as biological invasions progress. White boxes indicate typical management actions necessary to achieve each management objective. Post-establishment management actions are not shown under panel B since these are generally not achievable in these systems. In a management context, the first detection (introduction point), the lag phase (see **Glossary**) and the exponential spread phases are important points to design an early detection and rapid response management plan. This figure is conceptual, and the curves do not represent actual population dynamics of invasive alien species.

1.5 SOCIOECOLOGICAL CONTEXT

Increasing attention has been given to understanding the context dependency of biological invasions (Pyšek, Bacher, *et al.*, 2020; Sapsford *et al.*, 2020). Thus, the outcome of each biological invasion not only depends on the propagule pressure and traits of the species invading, but on the recipient ecosystem and its defining parameters within a specific time span and a specific spatial scale (Pauchard & Shea, 2006; **section 1.3.2**; **Figure 1.9**). This context dependency goes beyond ecological parameters as it is at least partly determined by human culture, incorporating behaviour, government policies and regulations, and other social components, including social differentiation and, at times, violent conflict (**Figure 1.9**; Howard, 2019; Kelsch *et al.*, 2020).

Modelling and predicting the spread and potential impacts of invasive alien species on biodiversity and human health and well-being are widely seen as critical to better curtail the harm they can cause to ecosystems and human communities (**Chapter 4**, **section 4.7.1**, and **Chapter 5**, **section 5.6.3.2**). Although there have been considerable advances in this regard, increasingly, scientists are recognizing the inherent difficulties of forecasting these processes in complex socioecological systems (Lenzner *et al.*, 2019). There are several reasons why this remains the case, despite progress in both the natural and social sciences in the study of biological invasions.

Invasive alien species respond to multiple natural and anthropogenic drivers (**Chapter 3**), which can also have synergistic effects on the outcomes of biological invasions. Pörtner *et al.* (2021) highlight the importance of recognizing the complex and multiple connections between climate and other drivers of change in nature. For example, positive feedback loops between plant invasions and more intense and frequent fires **(Box 1.4)** associated with climate change can completely shift fire regimes (Brooks *et al.*, 2004). The sphere of social interactions and human behaviour increases

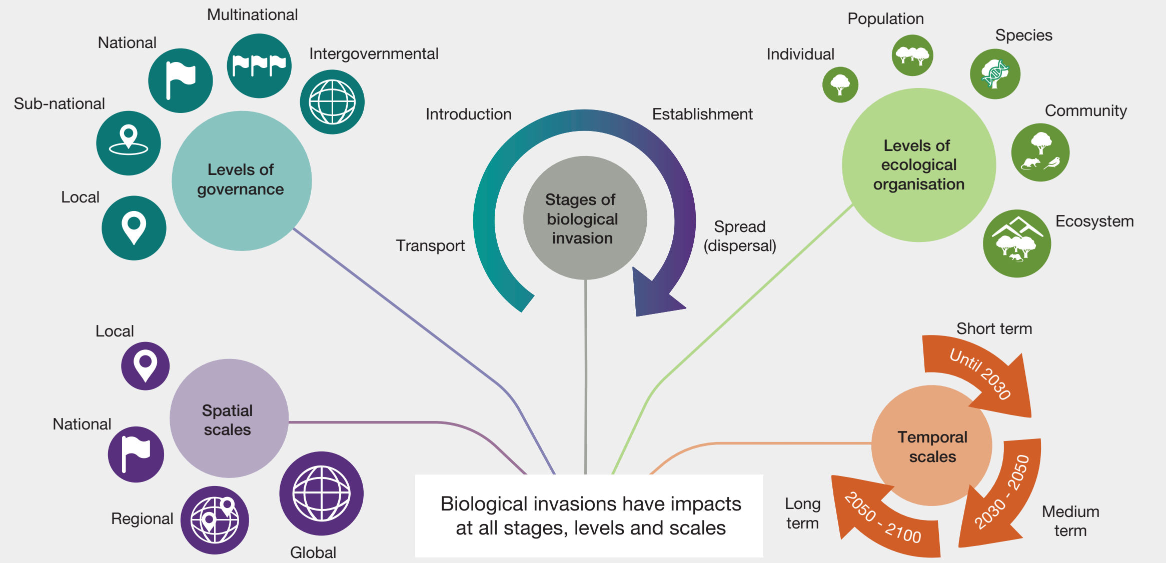


Figure 1.9 **Context dependency in biological invasions across multiple spatial and temporal scales, and governance and ecological levels.**

Underlying processes span various spatial (bottom-left: local to global) and temporal scales (bottom-right: short to long-term). Impacts of invasive alien species on nature, nature's contributions to people and good quality of life also vary across temporal and spatial scales and may differentially affect each level of ecological organization (top-right: from individuals to ecosystems). For some invasive alien species, the impacts are immediate and continue into the long-term (e.g., fast-spreading pathogens such as Zika virus, or fast-spreading predators such as lionfish) while for others there may be a considerable time lags, spanning decades in some cases, before the impacts are apparent (e.g., many invasive alien trees, see Kowarik, 1995). Some invasive alien species have local impacts (e.g., *Carassius auratus* (goldfish) released into small ponds by pet owners) while others impact globally (e.g., *Batrachochytrium dendrobatidis* (chytrid fungus)); and while many invasive alien species have impacts at the individual, population, or community level, others adversely impact entire ecosystems (e.g., eucalyptus and pine trees transforming native grasslands into shrub or wood land). Finally, different levels of governance (top-left: from local to inter-government) affect how biological invasions progress and are managed (e.g., local governance of invasive alien species may differ from national or international policies).

the complexity of mitigation efforts, which can be very difficult to communicate to policy- and decision-makers, to a wide variety of stakeholders, and to Indigenous Peoples and local communities. The effects of human caused fires (i.e., ignition) associated with a particular cultural behaviour have the potential to accelerate fire regime changes, and complicate management decisions alongside the outcomes from biotic and abiotic modelling. As another example, many aquatic invasive alien species are spread through recreational boating and if people who engage in this activity are unaware of the need to practice hull cleaning, and of the damage that invasive alien species can inflict on other recreational pastimes, they will be unlikely to take part in mitigation efforts.

Human responses to the threats posed by invasive alien species, including the introduction of alien species to achieve biological control and the use of chemicals or other agents in eradication programmes, can also affect the possibility of future biological invasions and the range of management responses and policy choices (**Chapters 5** and **6**). If people have begun to adapt to the presence of invasive alien species in a way that benefits them, then efforts to eradicate these species may not be seen as acceptable by some stakeholders (Howard, 2019), and there may also be resistance, on ethical grounds, to management methods that involve lethal responses.

Understanding the process of biological invasions within the context of varying spatial and temporal scales is important but can be challenging, because mechanisms underpinning the patterns are influenced by scale and the peculiarities of the phenomena being studied (Pauchard & Shea, 2006; Sapsford *et al.*, 2020). While patterns of biological invasions have now been documented at multiple spatial and temporal scales (**Chapter 2**), most studies have explored the mechanisms behind biological invasions only at small spatial scales because of the difficulties in experimental design and replicability. Furthermore, most mechanistic studies only look at short periods of time (i.e., a few years). Thus, there is still a critical gap in understanding the process of biological invasion over a range of scales. Simple scaling up is of limited value because processes and mechanisms vary at different scales and changes over time are rarely linear (Kowarik, 1995; Levin, 1992). However, in the last two decades and because of the accumulation of extensive observational datasets and the development of new analytical tools (Sagarin & Pauchard, 2012), macroecological studies are filling some of these gaps. It is now possible to consider invasive alien species on large temporal and spatial scales and therefore link patterns to processes and reveal underpinning mechanisms more robustly than was previously possible (e.g., Seebens *et al.*, 2015, 2021; **Chapter 2**). Indeed, the first estimates of future alien species projections, based on long-term alien species trends, are now available (Seebens *et al.*, 2021), indicating that past trends of invasive alien species will continue to accelerate for many taxonomic groups and regions. Multiscale solutions can help to address the threats posed to the natural world by multiple drivers of change in nature (Bonebrake *et al.*, 2019).

1.5.1 Characterizing stakeholders and biological invasion stages

Invasive alien species can variously affect, and be affected by, different categories of stakeholders across the stages of the biological invasion process **(Figure 1.10)**. A stakeholder refers both to those people who have the capacity to affect (influence) or are affected by (have interests in) biological invasion processes, outcomes, and policies. The IPBES invasive alien species assessment identifies three groups of stakeholders in relation to stages of the biological invasion process. They include "influencing stakeholders", who influence biological invasion processes, management or policies; "affected stakeholders", who are affected by biological invasions as "winners" or "losers"; and "contributing stakeholders" **(Figure 1.10)**, who contribute directly or indirectly to biological invasions without necessarily being influential or affected (Dandy *et al.*, 2017). Such groups are not mutually exclusive – both individuals and organizations can belong to several of these categories **(Figure 1.10)**.

Within the "influencing" and "affected" stakeholder groups, Dandy *et al.* (2017) identify several categories of stakeholders, described in **Table 1.1**.

1.5.2 Perceptions and values

Social and cultural dimensions of biological invasions encompass people's awareness, perceptions, values, attitudes, and interests **(Table 1.2)**. The study of these dimensions helps to better understand social conflicts, engagement and action or inaction throughout the biological invasion process described in **section 1.4**, and particularly in the context of the management of biological invasions and control of invasive alien species (Estévez *et al.*, 2015; Kueffer & Kull, 2017; Novoa *et al.*, 2017; Shackleton, Richardson, *et al.*, 2019). Some key literature from the environmental humanities has been critical in drawing attention to the entanglement of the ecological context and cultural values in biological invasions (Frawley & McCalman, 2014; Head, 2017; Tassin & Kull, 2015) and in showing that management of biological invasions depends on human decision making and behavioural change for success (Head *et al.*, 2005; McNeely, 2001).

Research activity on the social and cultural dimensions of biological invasions is slowly accelerating but is still in

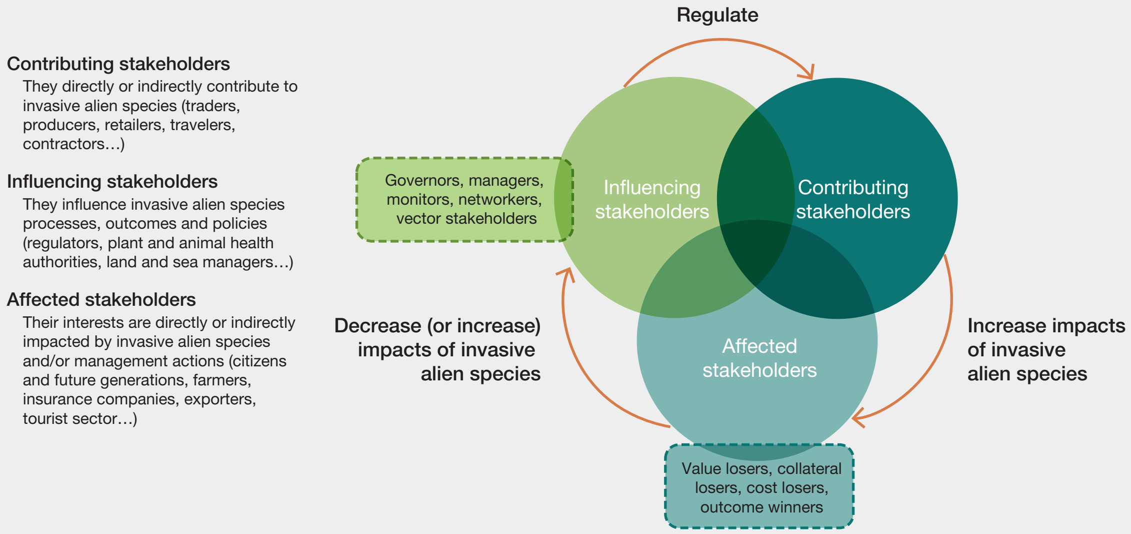


Figure 1 10 **Involvement of different stakeholder groups in the context of biological invasions.**

Table 1 1 **Groups and categories of stakeholders considered in the IPBES invasive alien species assessment.**

For a full description of the Stakeholder categories, please consult **Supplementary material 1.1.**

Stakeholder group	Stakeholder category	Description
Influencing stakeholders	Vector-stakeholders	Individuals or organizations whose activities, intentionally or unintentionally transport, introduce and/or spread invasive alien species
	Governors	Individuals or organizations who set formal and informal rules or establish norms that guide and drive management of biological invasions and adaptation, including prevention across all stages of the biological invasion process
	Monitors	Individuals or organizations who predict, identify, detect, conduct surveillance of and share information on invasive alien species across all stages of the biological invasion process
	Managers	Individuals or organizations who undertake “on-the-ground” responses to biological invasions across all stages of the biological invasion process
	Networkers	Individuals or organizations who disseminate information and key messages between actors relevant to biological invasions management, connecting other stakeholders with differing perspectives and operating at different scales
Affected stakeholders	Value losers	Individuals or organizations for whom nature’s contributions to people and good quality of life are reduced by invasive alien species or by management responses across all stages of the biological invasion process
	Cost losers	Individuals or organizations who bear the direct economic costs of responding to invasive alien species, such as paying for labour and materials required for eradication or containment, or for information dissemination across all stages of the biological invasion process. These direct costs can be incurred in addition to the loss of existing value (i.e., cost losers may often also be value losers)
	Collateral losers	Individuals or organizations who lose value indirectly as a consequence of the adverse impacts of invasive alien species or their management across all stages of the biological invasion process
	Outcome winners	Individuals or organizations for whom nature’s contributions to people and good quality of life are increased by invasive alien species or by their management responses across all stages of the biological invasion process. In some cases, invasive alien species provide additional nature’s contributions to people, in other cases, these stakeholders are able to turn harm into benefit
Contributing stakeholders	Individuals or organizations who directly or indirectly contribute to biological invasions	

Table 1 2 **Primary and underlying factors that shape people's perceptions of invasive alien species.**

Updated from Shackleton, Richardson, *et al.* (2019).

Primary factors driving perceptions of invasive alien species	Example sub-categories
Individual(s)	Demographic characteristics (gender, education, job, etc.) Experience of species and effects Knowledge systems Sense of place Social relationships and group membership Individual values and beliefs Livelihood strategies
Species	Introduction and species status (invasion status) Residence time Species traits Taxonomic/functional group Species charisma
Effects/Impacts (potential and realized) (beneficial and detrimental)	Economic Ecological Social, religious, and cultural Food security
Socio-cultural contexts	Land tenure system Management history Public and media discourse Socio-economic development Social and cultural institutions and value systems Relationship to the land Social memory Language used Livelihoods
Landscape context	Availability of alternative resources (e.g., from native species) Ecosystem type Land use and cover Landscape beauty/scenery or attractiveness Management history Ecosystem services
Institutional, governance and policy context	Historical processes Institutional frameworks International agreements Legislation, regulation, and enforcement Policy and governance strategy Scientific knowledge and understanding Power and responsibility

its infancy (Kapitza *et al.*, 2019; Vaz *et al.*, 2017). There have been important contributions to the understanding of biological invasions from the humanities **(Box 1.8)** and social sciences. However, a review of studies on biological invasions published between 1950 and 2014 revealed that contributions from the social sciences were limited to less than five per cent and that up to the 1990s interdisciplinary collaborations were largely confined to interactions between ecological and environmental sciences (Vaz *et al.*, 2017).

Kapitza *et al.* (2019) conducted a systematic review of studies on social perceptions of invasive alien species published before 2016. While the scope of this study was limited to the perception of invasive alien species themselves (thus excluding studies on perceptions of control or management of invasive alien species) it does reveal some important insights. First, most studies investigated perceptions of the general public (79 per cent), followed by decision-makers' (35 per cent) and scientists' (23 per cent) perspectives. Second, these studies reported a frequent use of quantitative methods using questionnaires, while only 14 per cent of the studies used qualitative methods such as interviews. Arguably, this indirectly led to a bias towards measuring perceived detrimental impacts of invasive alien species as these were more commonly included as items in questionnaires than the benefits of invasive alien species. Third, there were large biases in taxonomy (more than half of the studies (58 per cent) focused on plants), ecosystems (the majority of the studies (78 per cent) focused on terrestrial ecosystems), and geographical region (more than half of the studies were conducted in either North America (32 per cent) or Europe (28 per cent). This systematic review demonstrates the difficulty of ascertaining a clear picture of social perceptions

Box 1 8 Contributions to understanding of biological invasions from historical studies.

Since the emergence of the field of invasion biology in the 1980s, ecologists have increasingly recognized that the study of biological invasions involves significant ethical and cultural considerations that fall outside the purview of the biological sciences (Frawley & McCalman, 2014; Simberloff & Rejmanek, 2011). Historians have contributed to this research in three key ways: 1) by identifying the historical drivers of species migration; 2) by describing the emergence of narratives of biological invasion in scientific discourse and the impacts of invasive alien species control programmes; and 3) by deconstructing the language of prevalent biological invasions frameworks. They have shown that although species have always migrated across ecosystems, species movement accelerated from the eighteenth century onwards due to the mobilization of global agriculture, the extraction of biological matter for "exotic" horticulture, and land-use change (K. Thomas, 1984; Robbins, 2002; Ritvo, 2014; Bewell, 2017). Historians have described this advent of species movement "the Columbian exchange" (Crosby, 1972) and "ecological imperialism" (Crosby, 1986); few would disagree that the spread of commercial trade has been and continues to be the main driver facilitating species' introductions, including those now driven by climate change.

Legislation permitting the widespread control of certain plants and animals, unintentionally imported to colonial plantations, that had negative impacts on crops date back to the late eighteenth century. However, it wasn't until the late nineteenth century that some alien species were described as invasive. Historians have pointed to Charles Darwin, T.H. Huxley, his grandson Julian Huxley, and Charles Elton as key figures in the articulation of invasive alien species as a subject of scientific interest. This emergent narrative of biological invasion has been associated with xenophobia, successive wars, the start of the collapse of European empires, and early science fiction that addressed themes of alien invasion and scientific attempts to control it (Alt, 2010; Hovanec, 2018; Chang, 2019).

Historians and geographers have argued that neither "invasive" nor "native" are stable characteristics but are rather narratives of behaviours and interactions between species in ever-changing bio-cultural environments (Cronon, 1992; Smout, 2003; Frawley & McCalman, 2014). Such narratives often change over time (Hobbs *et al.*, 2006; Pawson & Christensen, 2014; Rangan & Kull, 2009; Ritvo, 2014). Some argue that "invasive" implies the previous existence of a static biota free from alien species when no such past exists (Rotherham & Lambert, 2013; Ritvo, 2014). Others have analysed the theory of "shifting baselines" — the way that each generation, without considering historical factors, bases science and policy decision-making around their own ecological circumstances (Dizard, 2010; Pauly, 1995; Vera, 2010).

Several critical studies have addressed the power of narratives about biological invasions in driving responses to changing environments such as eradication programmes (Smout, 2003; Trigger, 2008), and suggest that such stark binaries obscure the dynamism of changing environments (Head & Muir, 2004; Beinart & Wotshela, 2003; C. D. Thomas, 2017; Shah, 2020), including biodiversity gains and cultural losses. Failures to consider the diversity of rights-holders and stakeholders when addressing anthropogenic drivers of species loss in the past have enabled the continuation of colonial science in conservation decision-making (Grove, 1996; Griffiths & Robin, 1997; Caluya, 2014). Some historians urge that there is a need to emphasize the role of class and race in order to avoid deepening global inequalities (Nixon, 2011; Moore, 2016; Caluya, 2014). Researchers across the humanities are nevertheless in agreement that to solve the current and future interconnected problems of the global environmental crisis, we need to understand the complex interactions of ecologies, cultures, and societies of the past.

of biological invasions, despite their importance to the IPBES invasive alien species assessment.

An important aspect of perception is public awareness of invasive alien species. Public awareness is notoriously difficult to measure, but it is fundamental if preventive regimes (see **Glossary**) are to be adopted within communities. Schelhas *et al.* (2021) conducted an extensive review of public awareness and derived four important conclusions:

1. Knowledge of public awareness of invasive alien species is still quite limited and comes from either case study research or census studies. Case studies found that people are often generally aware of the existence of invasive alien species, but have limited knowledge about specific species, their impacts on biodiversity or the role of people in their introduction (e.g., García-Llorente *et al.*, 2008; Lindemann-Matthies, 2016; Verbrugge *et al.*, 2013, 2014). Findings from a survey on attitudes of citizens towards biodiversity show that, across Europe, introduced plants and animals are perceived as a lower threat to biodiversity compared to air and water pollution, human-made disasters, intensive farming, deforestation and over-fishing, climate change and conversion of natural areas to other uses (European Commission, 2013, 2015, 2019). However, in highly impacted locations, such as Hawai'i (Kalnicky, 2012) and in countries with a long history in plant and animal invasions, such as New Zealand (Hulme, 2020b), public interest and knowledge are often greater, as is support for management.

2. Invasive alien species are often viewed differently by the public than by scientists or policy makers. A mail

survey in the United States showed that members of the public ranked invasive alien species as 19 out of 24 ecological risk items, while professional risk assessors ranked them as ninth (Slimak & Dietz, 2006). A species' perceived harmfulness and human responsibility for its spread were the most important animating factors, while non-nativeness did not necessarily raise concerns (Qvenild *et al.*, 2014; Selge *et al.*, 2011). However, species' charisma (characteristics that positively affect the perceptions, attitudes, and behaviours of people towards them) can also have implications on public perceptions and consequently management interventions (Jarić *et al.*, 2020). Time also plays a role in shaping public perceptions, as people may be unaware of the origin of introduced species as they are regarded as normal or desirable in their natural surroundings (Genovart *et al.*, 2013) – this is sometimes also referred to as shifting baseline syndrome (Clavero, 2014).

3. It is suggested that the terminology employed to call attention to invasive alien species and their control should be chosen carefully (Clergeau & Nuñez, 2006; Janovsky & Larson, 2019; Larson, 2005; Verbrugge *et al.*, 2016). The use of metaphors or derogative language is common in both scientific and popular writing about biological invasions, but little is known about the effects on public values or opinions. How the issue-area is framed by officials, scientists, politicians, and other leaders will have an impact on subsequent policy development; biological invasions can be seen primarily as threats to biodiversity, national security, human health, trade, or even cultural homogeneity (Stoett, 2010).

4. Indigenous voices and values are under-represented in scholarly discourse about invasive alien species (e.g., Bhattacharyya & Larson, 2014). The IPBES invasive alien species assessment has attempted to be inclusive, but see Schelhas *et al.* (2021) for an elaborative view on the importance of considering Indigenous and local knowledge, unique cultural dimensions and engaging Indigenous Peoples and local communities in the management of biological invasions and the control of invasive alien species, using two examples from the United States to show how invasive alien species can either culturally impoverish or enrich Indigenous Peoples and local communities (see also Pfeiffer & Voeks, 2008). The social justice concerns related to Indigenous Peoples and local communities as they manage biological invasions should not be overlooked (Head & Atchison, 2015).

Perceptions of invasive alien species and support for management are thus influenced by a wide range of values (**Table 1.2**; **Boxes 1.9** and **1.10**; see also Carter *et al.*, (2021) who extend this overview with ethical considerations for including social perspectives in research planning and decision-making). Research in the past five years has become more diverse in terms of theoretical and methodological approaches, for example by analysing how socio-historical processes interact with biological invasions (Archibald *et al.*, 2020), developing "sense of place" as a concept to explain how place attachment can promote or impede action against invasive alien species, or reframing biological invasions as socioecological phenomena to enhance cross-fertilization across ecological sciences and social sciences (Gawith *et al.*, 2020; Vaz *et al.*, 2017). Encouragingly, collaborative knowledge platforms are being developed (e.g., Bennett & van Sittert, 2019; Udo *et al.*, 2019), but further efforts for realizing collaboration between natural and social sciences are much needed for a more holistic understanding of perceptions of invasive alien species and critical for developing adequate control and policy responses.

Box 1 9 **Human values and the invasive alien carp in North America.**

A group of invasive alien carps (cyprinid fishes) were brought from Eastern Asia to Arkansas, United States of America in the 1960s to serve as biological control agents in aquaculture ponds (Besek, 2019). Many escaped soon after their importation and have since been migrating up the Mississippi River watershed, adversely impacting both social and ecological systems along the way. Since the early 2000s, many stakeholders with an interest in the North American Great Lakes have been advocating for the construction of a hydrologic barrier to stop invasive alien carp from entering and impacting their fisheries. This proposed barrier, however, would drastically impact regional shipping and transportation, setting up a substantial political battle regarding how to best manage invasive alien carp spread. This contentious social context has significantly impacted the work of scientists trying to assess invasive alien carp migration, tying their work to local politics and human values in numerous ways. For instance, most scientists have refused to offer unqualified predictions about the future migration of invasive alien carp because the ecological processes involved are so complex, and many political actors have seized on this indeterminacy to publicly question science methodologies and laboratory techniques used to study invasive alien species. Some scientists have been requested to explain and defend their work in federal courtrooms. This heated political climate has in some ways given extra attention to detection techniques, improving their precision, but has also led many scientists to avoid working on invasive alien carp altogether.

Box 1 10 **Conceptual perspectives from the social sciences.**

Social science and humanities research on biological invasions has grown steadily since the 1990s (Vaz *et al.*, 2017). Some of this work addresses perceptions, attitudes, and behaviours with a perspective towards enabling management and control of invaders (Rotherham & Lambert, 2013; Shackleton, Richardson, *et al.*, 2019). Arguably, when social science is integrated with biological invasion science, it has followed an "ABC" framework, focusing primarily on attitudes, behaviour, and choice (Shove, 2010). Some researchers are leaning towards more explicitly "critical" approaches to biological invasion science (Head, 2017; Kull, 2018). By "critical", social scientists refer to approaches that question underlying processes and conceptual foundations, seeing knowledge as political and transformative.

Several factors inform a critical social science perspective. It is challenging to consider landscapes being invaded without looking at how they have been co-produced by humans in myriad ways (for instance, clearance, soil degradation and introductions), and in many cases the invasive alien species themselves (for instance, genetic selection for species that have been introduced). This focus shifts attention from dangerous invaders to human complicity in biological invasions (Kueffer, 2017). Second, the study of invasive alien species has a specific trajectory and social context that shapes the knowledge produced on biological invasions (Archibald *et al.*, 2020). The social-political context of the institutions that undertake biological invasion-related research and seek to manage biological invasions and control invasive alien species (state weed agencies, land managers), is relevant, as this determines the voices and knowledge systems that are heard. The IPBES conceptual framework is attentive to the need to examine a variety of knowledge systems (Díaz *et al.*, 2015). A third necessity is to investigate how knowledge about invasive alien species is used and implemented, and what the consequences are for people and landscapes (Kull, 2018). The establishment of lists of high risk invasive alien species, for quarantine systems, or for community weed-pulling days; sending rangers out to spray herbicides on invasive alien plants or lay poison traps for invasive alien animals; establishing major public works policies like South Africa's "Working for Water" programme – each of these actions has knock-on effects, creates winners and losers, and creates ripples in the system that are not entirely predictable nor agreed to by all parties (Atchison & Head, 2013; Bach *et al.*, 2019; Fall, 2013; Gallardo *et al.*, 2019; Head *et al.*, 2015).

1.5.3 Ethics and invasive alien species

The management of invasive alien species, in particular sentient animals, raises multiple ethical debates with regards to animal welfare and rights, and this is considered an under-addressed animal welfare issue in conservation (Carter *et al.*, 2021; Doherty & Russell, 2019; Hampton & Hyndman, 2019; **Chapter 5**, **section 5.6.2.1**).

There are philosophical differences between proponents of animal rights, who focus on the individual animal, and those who focus on conservation at a species or ecosystem level, with the former having an increasing influence on public opinion and legislation. The extension of legal rights to animals and nature imposes moral and legal limits on acceptable human uses of the environment, and if the legal personality (**Glossary**) of both ecosystems and individual animals is acknowledged, the interests of individual animals may conflict with interests of individual species, as can be the case with native and invasive alien species (Futhazar, 2020). Arguably, the rights of native species to exist need to be respected (hence the importance of prevention and adapting the precautionary principle) but once an invasive alien species is established, the picture is more complicated.

Deciding whether and how to control invasive alien species involves analysing risks, and considering international consensus principles for ethical wildlife control which are informed by social and cultural values in addition to scientific, technical, and practical information. As discussed above, there is a diverse range of perceptions of invasive alien species, both positive and negative (Shackleton, Richardson, *et al.*, 2019). Moral dilemmas posed by controlling invasive alien species can involve subjective judgements about the perceived ecological value of protected species *versus* the lack of importance of invasive alien species (Mankad *et al.*, 2019) or indeed the charisma of one species compared to another (Jarić *et al.*, 2020).

Different invasive alien species management methods can raise different ethical debates. Genome editing can pose ethical questions because of concerns about the risks and unknown consequences of releasing genetically modified plants or animals into the wild (**Chapter 5**, **section 5.4.4.2**) (Bertolino, 2020). Gene suppression-drives may pose risks to global populations of invasive alien species and so are being considered with caution (Thresher, 2020). There are several reports outlining the risks and opportunities of these technologies (**Chapter 5**, **section 5.4.4.2**; Invasive Species Advisory Committee (ISAC), 2017; Redford *et al.*, 2019). Biological control can pose potential social and environmental risks, but often brings benefits (Müller-Schärer *et al.*, 2020; Thomas & Willis, 1998), and evokes a normative debate (Mankad *et al.*, 2019). It is relevant to consider social values and emotional and cultural associations, in addition to stakeholder preferences, humaneness and effectiveness, when managing invasive alien species (Mankad *et al.*, 2019).

Lethal management methods can be particularly controversial and a framework for assessing the success and sustainability of a particular management decision that takes into account ecology, economic and ethics has been proposed (Warburton & Anderson, 2018). Prevention is often the "preferred option for managers and desirable and philosophically acceptable to animal rights advocates" (Perry & Perry, 2008). Furthermore, proponents of compassionate conservation state that humans should do no harm and consider that individual animals matter.

Given the range of values and management options, there are unique conceptual and governance challenges associated with invasive alien species (Stoett, 2007). The language used to describe invasive alien species has sometimes been labelled as nativist (Gbedomon *et al.*, 2020), and is predominantly negative. Inglis (2020) states that; "the invasive discourse is couched in language which immediately prejudices people against the animals. This leads to the killing of these animals being viewed as both morally acceptable and indeed necessary." Nevertheless, Shackelford *et al.* (2013) suggest finding middle-ground in the native/non-native debate that recognizes the merits of both sides when assessing management options. Furthermore, there is no globally accepted definition of animal welfare and interpretation of the concept of animal welfare evolves with advances in our understanding of animals (Dawkins, 2017; Harrop, 2013; Mellor *et al.*, 2020; White, 2013).

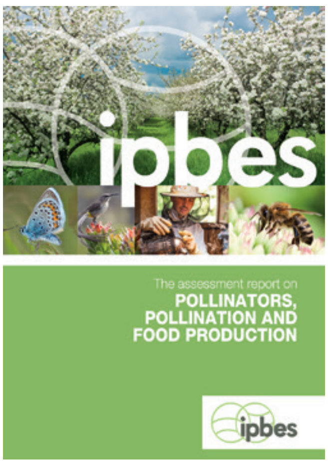
An eighteenth Sustainable Development Goal on animal health, welfare and rights has been suggested to ameliorate trade-offs between animal welfare and sustainability, with the management of invasive alien species noted as an example (Visseren-Hamakers, 2020). Accordingly, as discussed in **Chapter 6**, balancing values across multiple and interrelated stakeholder groups is an important consideration within invasive alien species management (Carter *et al.*, 2021).

1.6 CONCEPTUAL BASIS FOR THE INVASIVE ALIEN SPECIES ASSESSMENT

IPBES assessments aim to identify policy-relevant findings for decision-making in government, the private sector and civil society by synthesizing and critically evaluating peer-reviewed scientific literature, grey literature, and other available knowledge, such as Indigenous and local knowledge. Assessments do not generate new data, but seek to create new understanding through summary, sorting and synthesis using different methods to manage complexity.

The IPBES invasive alien species assessment builds upon several IPBES assessments, which include

Box 1 11 **Biological invasions and pollination processes.**



The IPBES Assessment Report on Pollinators, Pollination and Food Production (IPBES, 2016b) considered the outcomes of biological invasions on pollinator populations, diversity, network structure and pollination processes and confirmed that ecological and evolutionary contexts are important. Although predicting the consequences of the arrival of an invasive alien species within a pollinator network is difficult, because of the ecological complexity inherent with multiple interacting species, it is apparent that the trophic position (plant/herbivore/pollinator/predator) and degree of specialization of an invasive alien species can be informative. Invasive alien species can alter the function, structure and stability of plant-pollinator networks with adverse impacts on specific native pollinator species and, sometimes, reductions in overall pollinator abundance or diversity (Vilà *et al.*, 2009). In native pollination networks dominated by generalist plants and pollinators, invasive alien plant species are often readily integrated. Consequently, networks including alien plants are characterized by increased plant and pollinator richness and high values of nestedness (Stouffer *et al.*, 2014). As an example, alien species (plants and pollinators) comprised 56 percent of the total number of interactions within pollination networks on the Galápagos Islands. Alien insects within these pollination networks linked mostly to generalist plant species resulting in increased nestedness and network stability (Traveset *et al.*, 2015). Such changes to the community structure increase network cohesiveness but disrupt native ecological interactions (Traveset *et al.*, 2015). The impacts of invasive alien species on pollinators and pollination are likely to be further exacerbated when coupled with other threats including wildlife diseases, climate or land-use change (González-Varo *et al.*, 2013; Schweiger *et al.*, 2010; Sunny *et al.*, 2015; Vanbergen & Initiative, 2013).

thematic assessments of Pollinators, Pollination and Food Production (IPBES, 2016b; **Box 1.11**), Land Degradation and Restoration (IPBES, 2018c); Sustainable Use of Wild Species (IPBES, 2022c); Methodological Assessments of Scenarios and Models of Biodiversity and Ecosystem Services (IPBES, 2016c), and of the Diverse Values and Valuation of Nature (IPBES, 2022a); four regional assessments of Biodiversity and Ecosystem Services (IPBES, 2018d, 2018e, 2018f, 2018g); and the Global Assessment of Biodiversity and Ecosystem Services (IPBES, 2019).

1.6.1 The IPBES conceptual framework and its use in the invasive alien species assessment

The IPBES conceptual framework[4] aims to facilitate interdisciplinary collaboration and science-policy dialogues (Díaz *et al.*, 2015). It explicitly considers diverse disciplines, different stakeholders and Indigenous Peoples and local communities (**section 1.5.2**), and several knowledge systems (natural sciences, social sciences and humanities, Indigenous, local and practitioners' knowledge).

4. A full description of the IPBES conceptual framework, and associated definitions, is available in **Supplementary material 1.2**.

The IPBES conceptual framework includes six interlinked elements constituting a socioecological system that operates at various scales in time and space: nature; nature's contributions to people; anthropogenic assets; institutions and governance systems and other indirect drivers of change; direct drivers of change; and good quality of life.

The IPBES invasive alien species assessment falls within the IPBES conceptual framework, and uses it to understand how the major threat posed by invasive alien species can be reduced while those that are considered important components of nature and nature's contributions to people can be maintained in order to improve good quality of life. The assessment recognizes the importance of integrating this knowledge in the broader context of global change. By superimposing the specificities of the assessment over the IPBES conceptual framework, **Figure 1.11** shows the interactions between invasive alien species and the other elements of the IPBES conceptual framework. All these relationships are dynamic, changing over time, and different scenarios (i.e., trajectories for each component) are likely to lead to different outcomes. Socioecological contexts, including public awareness and stakeholder engagement levels, can also change according to the spatial scale under consideration (i.e., local, regional, global), thus affecting how invasive alien species are perceived and managed.

Box 1 12 Nature's contributions to people.

Nature's contributions to people are an integral part of the IPBES conceptual framework **(Figure 1.11)** and represent all the contributions, both positive and negative, of living nature (i.e., diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to the quality of life for people (Díaz *et al.*, 2018). Beneficial contributions from nature include such things as food provision, water purification, flood control, and artistic inspiration, whereas detrimental contributions include transmission of disease, particularly those affecting animal, plant, and human health **(Box 1.14)**, and other ways in which harm to people or their assets or community stability/resilience may occur as a consequence of invasive alien species. Many of nature's contributions to people may be perceived as beneficial or detrimental depending on the cultural, temporal, or spatial context (Díaz *et al.*, 2018; **sections 1.5.2**, **1.5.3**; **Chapter 4**, **section 4.1.3**). The concept of nature's contributions to people addresses the need to recognize the cultural and spiritual impacts of biodiversity, in ways that are not restricted to a discrete cultural ecosystem services category, but instead encompass diverse world views of human-nature relations (Mace, 2014).

The IPBES invasive alien species assessment adopts the 18 categories identified by IPBES for reporting nature's contributions to people (Díaz *et al.*, 2018). These 18 categories of nature's contributions to people are organized into three partially overlapping groups, according to the type of contribution they make to people's quality of life **(Figure 1.12)**:

Material nature's contributions to people: substances, objects, or other material elements from nature that directly sustain people's physical existence and material assets. They are typically physically consumed in the process of being experienced.

Non-material nature's contributions to people: nature's effects on subjective or psychological aspects underpinning people's quality of life, both individually and collectively.

Nature's regulating contributions to people: functional and structural aspects of organisms and ecosystems that modify the environmental conditions experienced by people, and/or sustain and/or regulate the generation of material and non-material contributions.

Box 1 12

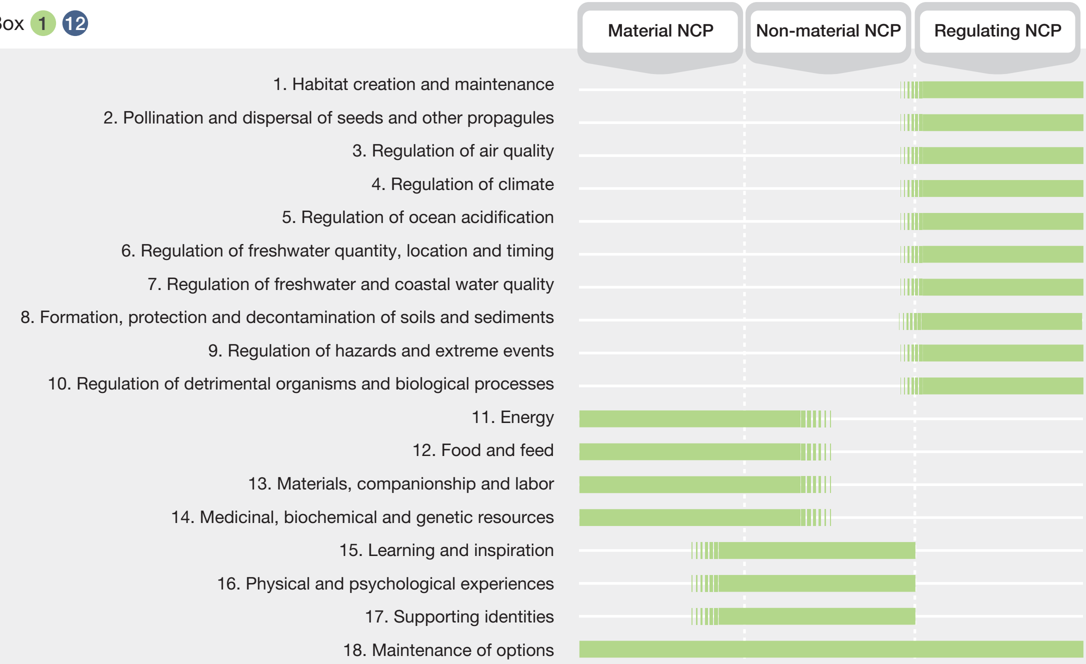
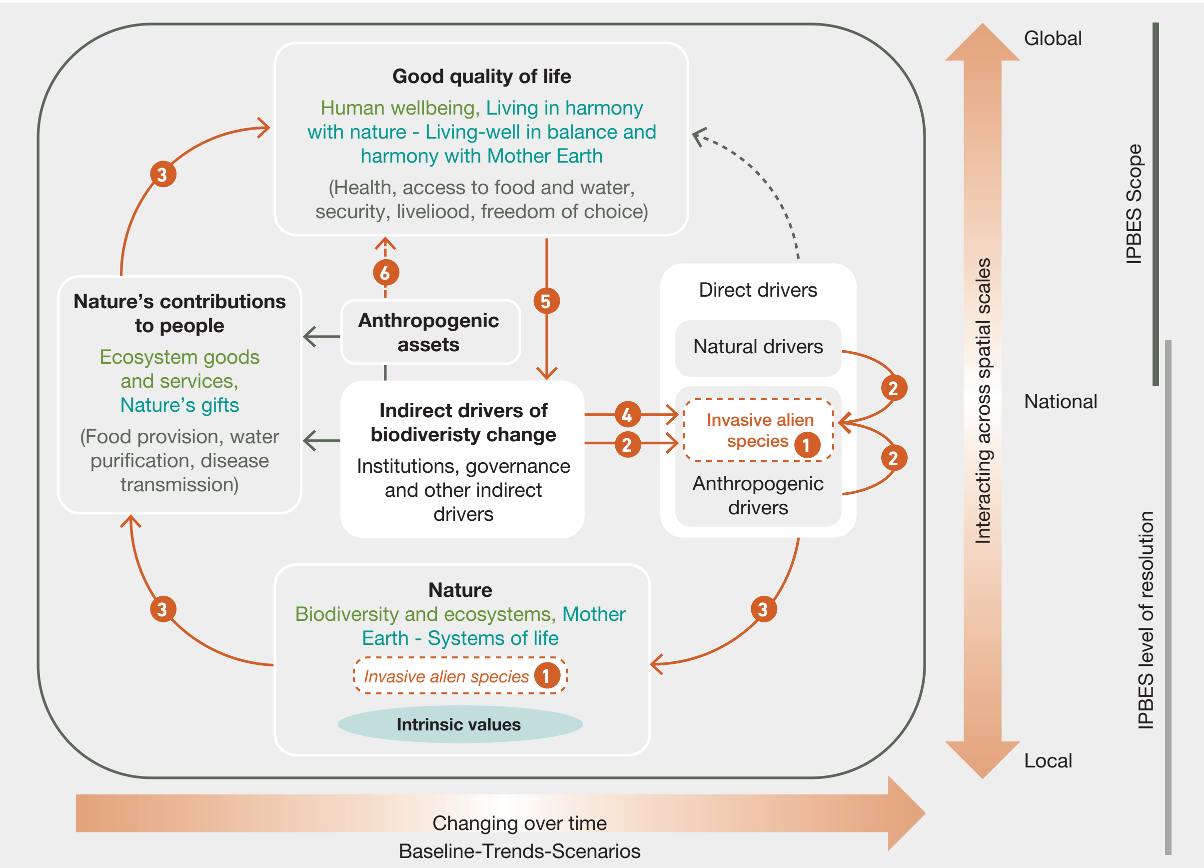


Figure 1 12 **Categories of nature's contributions to people (NCP).**

From Brondizio *et al.* (2019), https://doi.org/10.5281/zenodo.3831852, under license CC BY 4.0. These categories are used in this assessment of invasive alien species.



1 **Status and trends of invasive alien species:** invasive alien species are one of five major direct drivers of change, and at the same time are part of nature. This assessment captures both aspects, with their dynamics being addressed in **Chapter 2**.

2 **Synergies and interactions of invasive alien species with other drivers of change in nature:** the transport, introduction, establishment and spread of invasive alien species are facilitated, modified and amplified through interactions and synergies with other direct and indirect drivers of change in nature (e.g., climate change, economic drivers) as well as by natural hazards and biodiversity loss (addressed in **Chapter 3**).

3 **Impacts of invasive alien species on nature, nature's contributions to people and good quality of life:** invasive alien species impact nature in diverse ways, and often in ways that interact with other drivers of change in non-linear ways (synergistic, antagonistic) (addressed in **Chapter 4**). Changes to nature, including in ecosystem functions, underpin changes to nature's contributions to people (see **Box 1.12**), which can affect society in detrimental or, in some cases, beneficial ways (addressed in **Chapter 4**). The effects of invasive alien species on people and good quality of life (**section 1.6.7.2**) can be direct or through other components of the ecosystems (e.g., human health may be affected by parasites and contagious emergent diseases) (addressed in **Chapter 4**).

4 **Responses to biological invasions:** institutions, governance and other societal indirect drivers of change in nature can respond to biological invasions through direct management measures, including prevention and adaptation, restoration and policies (addressed in **Chapters 5** and **6**).

5 **Influence of people on responses to invasive alien species:** biological invasions' management and policies are driven by how people perceive and act in response to the threat of invasive alien species (addressed in **Chapters 1**, **5** and **6**).

6 **Adaptation to invasive alien species:** society can also adapt to invasive alien species and thus mitigate their adverse impacts on good quality of life; for example, invasive alien species can become new sources of food security (addressed in **Chapters 5** and **6**).

Figure 1 11 **The IPBES conceptual framework adapted to the IPBES invasive alien species assessment.**

Interactions amongst the components of the IPBES conceptual framework that are relevant to biological invasions are indicated in numbered arrows (boxes, arrows and numbers), with detailed descriptions provided in the lower panel of the figure. Unnumbered arrows represent the relationships between different components of the IPBES conceptual framework as defined in Díaz *et al.* (2015), that are not studied in this assessment. Adapted from Díaz *et al.* (2015), https://doi.org/10.1016/j.cosust.2014.11.002, under license CC BY-NC-SA 3.0.

1.6.2 Literature review

The IPBES invasive alien species assessment's findings emerge from systematic and transparent evaluations of available evidence to date[5] combined with experts' inputs, taking into account different worldviews and knowledge systems. Existing evidence encompasses published scientific and grey literature, including Indigenous and local knowledge, government publications, policy documents and briefs, technical reports and datasets, etc. This assessment also builds on previous IPBES assessments and other relevant global assessments such as the Global Biodiversity Outlook series, the United Nations Environment Programme (UNEP) Global Environment Outlook series, and the Millennium Ecosystem Assessment.

Authors were guided by the IPBES Data Management Policy (IPBES, 2020a), and the flexible protocol for systematic review that was first developed by the Global Assessment of Biodiversity and Ecosystem Services (Brondizio *et al.*, 2019; Collaboration for Environmental Evidence, 2013), which is critical to achieve scientific credibility and transparency of the assessment, following the FAIR (findable, accessible, interoperable, and reusable) data principles.

Authors sought to represent the most relevant and highest quality evidence, with the highest level of synthesis available as a priority; and provided supplemental material if necessary to fully cover and evaluate the topic, or to include the most up-to-date information. Methodologies and workflows for literature reviews usually include two practical steps: 1) concurrent database searches of different kinds of literature (e.g., peer reviewed and "grey" published literature, unpublished but openly available reports and databases) to minimize potential biases and 2) personal knowledge and experience of authors regarding key seminal resources or publications not appearing as an output from first step (if available).

Data and information have been compiled from many sources and domains spanning scales from local to global **(Figure 1.13)**. Throughout the chapters, following

5. The cut-off date for the inclusion of published sources was 15 December 2021, which corresponds to the start of the second external review (second draft of the chapters and first draft of the summary for policymakers). In line with IPBES procedures, additional citations were included passed this date when prompted by a comment made during the second external review (accessible at https://ipbes.net/ias) and when seen as relevant by experts.

extensive synthesis of available evidence, gaps in existing knowledge were revealed and documented with an overarching synthesis of gaps, and options for addressing them, provided within **Chapter 6**. The IPBES Regional Assessments of Biodiversity and Ecosystem Services all recognize gaps in data and information which are particularly pronounced in some regions and for many taxa (IPBES, 2018g, 2018f, 2018d, 2018e, 2018c). However, the growth in availability of datasets globally is encouraging (**Chapter 2**, **section 2.1.4**), although there remain lags in collating and sharing information on invasive alien species and consequently gaps in datasets across all regions.

The analysis of Indigenous Peoples and local communities' issues and knowledge also benefited from an "online call for contributions", which collected 30 references that were reviewed and selected to inform specific sections of the assessment. Three Indigenous and local knowledge dialogue workshops were also held throughout the timeframe of the assessment, which led to suggested literature and government reports being reviewed (**section 1.6.7.1**).

Authors documented their sources as well as their methodologies and workflows for literature reviews in data management reports, which are linked as footnotes, where appropriate. Across all chapters, references are cited within the text and the full reference is provided at the end of each chapter. The executive summaries of the chapters and the background text of the summary for policymakers include statements with traceability enclosed in curly brackets linking the statements to their underlying chapter subsections.

These systematic literature reviews, combined with expert-based critical opinions, are intended to enable the IPBES invasive alien species assessment to generate key findings and policy-relevant messages to support decision-makers in better understanding and tackling the complex issue of biological invasions and invasive alien species (**section 1.6.3**).

1.6.3 IPBES confidence framework

Confidence levels assist authors in the process of assessing and communicating the degree of uncertainty, or confidence, related to key findings. The evidence includes publications, data, theory, models and information **(Figure 1.14)** from multiple disciplines and knowledge

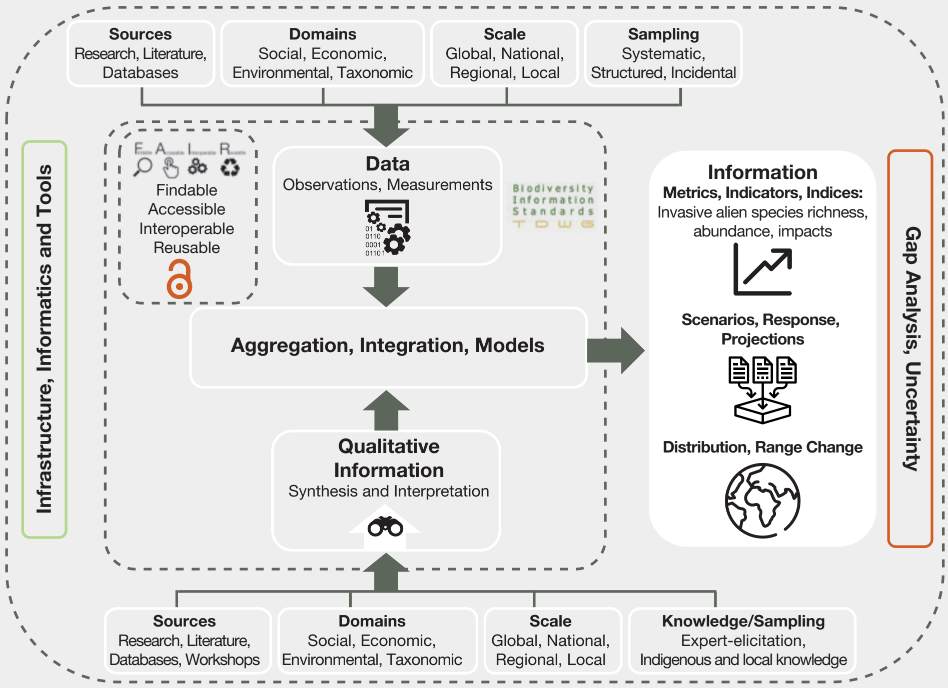


Figure 1 13 **Connections amongst types of evidence.**

Data and knowledge (**Chapter 2**) from many sources and domains spanning various scales and sampling techniques are combined to establish information in the form of metrics, indicators and indices which contributes knowledge on drivers (**Chapter 3**) and impacts (**Chapter 4**), ultimately informing management (**Chapter 5**) and future options (**Chapter 6**).

Figure 1 14 **The four-box model for the qualitative communication of confidence.**

Confidence increases towards the top-right corner as suggested by the increasing strength of shading. The four-box confidence framework developed for IPBES assessments is based on the quantity and quality of evidence assessed (x-axis) in combination with the level of agreement of experts using their judgment (y-axis). From IPBES (2016a), https://doi.org/10.5281/zenodo.2616458, under license CC BY 4.0.

systems. These confidence terms inform and communicate to decision-makers the degree of confidence that the assessment author teams associate to the key findings throughout the assessment and, importantly, highlight where further investigation is required to inform robust evidence-based decision making. Further details of the approach are documented in the IPBES Guide to the Production of Assessments (IPBES, 2018b).

The summary terms to describe the evidence are:

- **Well established:** There is a comprehensive meta-analysis or other syntheses/multiple independent studies that agree.
- **Established but incomplete:** There is general agreement although only a limited number of studies exist; there is no comprehensive synthesis, and/or the studies that exist address the question imprecisely.
- **Unresolved:** Multiple independent studies exist but their conclusions do not agree.
- **Inconclusive:** There is limited evidence and a recognition of major knowledge gaps.

1.6.4 IPBES regions and sub-regions

The IPBES invasive alien species assessment is global and encompasses alien species in terrestrial, freshwater and marine ecosystems across regions. It adopts the IPBES categorization of regions and sub-regions (**Figure 1.15**; IPBES technical support unit on knowledge and data, 2021) to structure its analysis (**Chapters 2, 3, 4, 5** and **6**). The IPBES technical support unit on knowledge and data (2021) also produced the dataset describing the IPBES regions and sub-regions and their corresponding countries or areas, in line with decision IPBES-3/1.

1.6.5 IPBES units of analysis

Each region and sub-region **(Figure 1.15)** are divided into multiple spatial units (biomes and ecosystems), spreading across borders. The invasive alien species assessment therefore adopts the 17 IPBES units of analysis (**Table 1.3**, see **Chapters 2, 3, 4, 5** and **6**) also used in previous IPBES assessments and defined by the IPBES Global Assessment (IPBES, 2019)[6] to support its analysis.

6. Definitions of the IPBES units of analysis available in **Supplementary material 1.3**

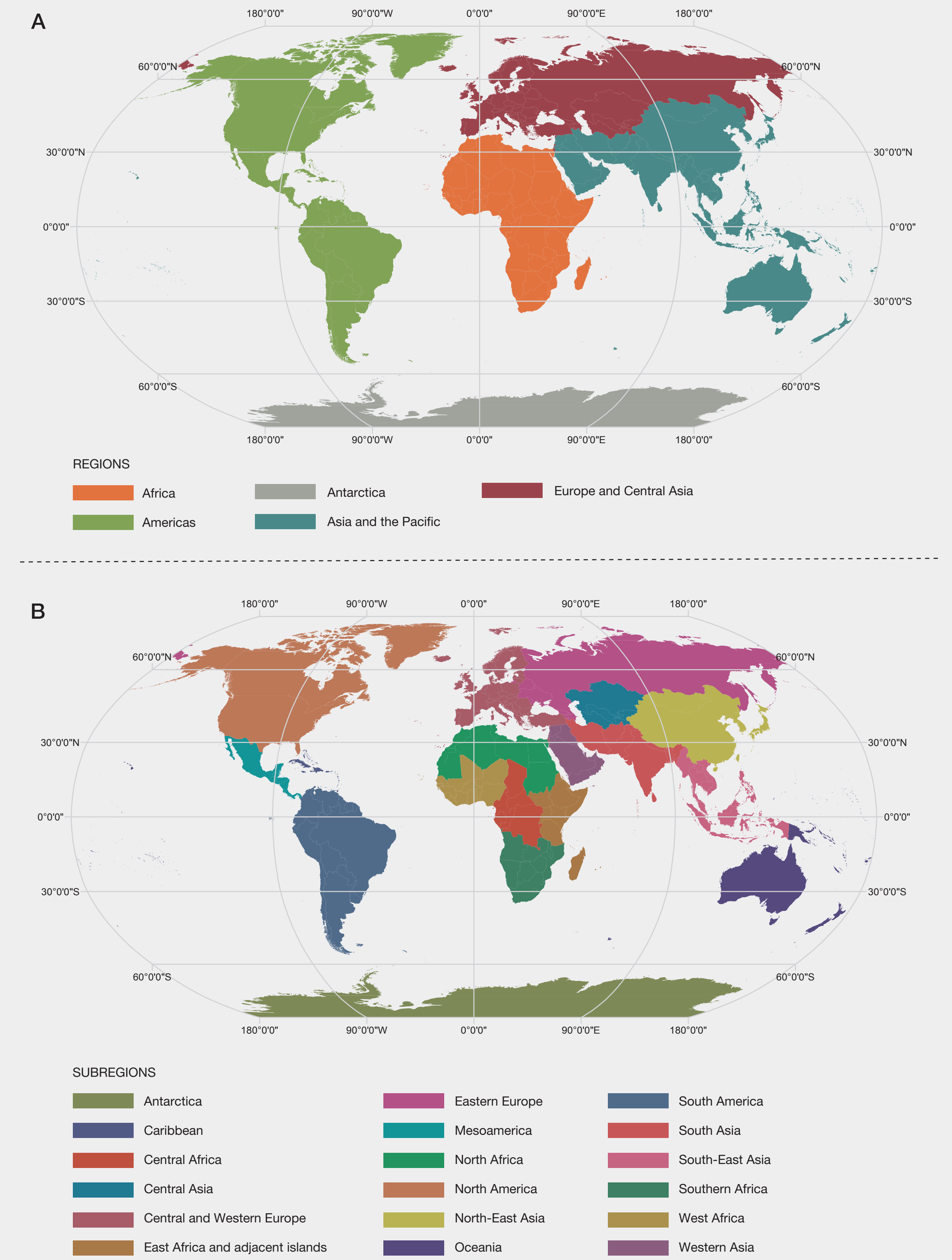


Figure 1 15 **IPBES regions (top) and sub-regions (bottom).**

Source: IPBES technical support unit on knowledge and data (2021), https://doi.org/10.5281/zenodo.5719431, under license CC BY 4.0.

Figure 1 16 **Photo montage of invasive alien species across regions and biomes.**

From top to bottom, left to right: *Pinus mugo* (mountain pine); *Passer domesticus* (house sparrow); *Rattus rattus* (black rat); *Vulpes vulpes* (red fox); *Vachellia nilotica* (gum arabic tree); *Aedes albopictus* (Asian tiger mosquito); *Undaria pinnatifida* (Asian kelp); *Lissachatina fulica* (giant African land snail); *Pontederia crassipes* (water hyacinth); *Solenopsis invicta* (red imported fire ant); *Procambarus clarkii* (red swamp crayfish); *Carcinus maenas* (European shore crab); *Caulerpa taxifolia* (killer algae); *Pterois miles* (lionfish); *Mustela vison* (American mink).

Table 1 3 **Examples of invasive alien species for each IPBES unit of analysis.**

The examples do not necessarily include the most widespread or harmful invasive alien species, but examples to provide representation of the diversity of species in each unit of analysis.

Unit	Biomes	Examples[7,8]- see Figure 1.16 for illustrations
1. Tropical and subtropical dry and humid forests	Terrestrial	*Cenchrus setaceus* (fountain grass) *Lissachatina fulica* (giant African land snail) *Cenchrus ciliaris* (buffel grass) *Homalodisca vitripennis* (glassy winged sharpshooter)
2. Temperate and boreal forests and woodlands	Terrestrial	*Lupinus polyphyllus* (garden lupin) *Lumbricus terrestris* (lob worm) *Pueraria montana* (kudzu) *Solenopsis invicta* (red imported fire ant)
3. Mediterranean forests, woodlands and scrub	Terrestrial	*Acacia longifolia* (golden wattle) *Pheidole megacephala* (big-headed ant) *Centaurea solstitialis* (yellow starthistle) *Aedes albopictus* (Asian tiger mosquito)
4. Tundra and high mountain habitats	Terrestrial	*Pinus mugo* (mountain pine) *Poa annua* (annual meadowgrass)
5. Tropical and subtropical savannas and grasslands	Terrestrial	*Prosopis glandulosa* (honey mesquite) *Felis catus* (cat) *Andropogon gayanus* (tambuki grass) *Bubalus bubalis* (Asian water buffalo)
6. Temperate grasslands	Terrestrial	*Pinus radiata* (radiata pine) *Pinus patula* (Mexican weeping pine) *Rattus rattus* (black rat) *Rattus norvegicus* (brown rat)
7. Deserts and xeric shrublands	Terrestrial	*Bromus tectorum* (downy brome) *Canis lupus dingo* (dingo) *Vachellia nilotica* (gum arabic tree) *Sus scrofa* (feral pig)
8. Wetlands – peatlands, mires, bogs	Freshwater	*Reynoutria japonica* (Japanese knotweed) *Mimosa pigra* (giant sensitive plant) *Procambarus clarkii* (red swamp crayfish) *Pomacea canaliculata* (golden apple snail)
9. Urban/Semi-urban	Human (anthrome)	*Parthenium hysterophorus* (parthenium weed) *Linepithema humile* (Argentine ant) *Lonicera tatarica* (Tatarian honeysuckle) *Aedes albopictus* (Asian tiger mosquito) *Passer domesticus* (house sparrow) *Sturnus vulgaris* (common starling) *Columba livia* (pigeons)

Table 1 3

Unit	Biomes	Examples[7,8]- see Figure 1.16 for illustrations
10. Cultivated areas (incl. cropping, intensive livestock farming, etc.)	Human (anthrome)	*Artemisia vulgaris* (mugwort) *Mustela vison* (American mink) *Acacia longifolia* (golden wattle) *Nosema bombi* (microsporidian parasite)
11. Cryosphere	Terrestrial, freshwater and marine	
12. Aquaculture areas	Human (anthrome)	*Undaria pinnatifida* (Asian kelp) *Magallana gigas* (Pacific oyster) *Carassius gibelio* (Prussian carp) *Pacifastacus leniusculus* (American signal crayfish)
13. Inland surface waters and water bodies/freshwater	Freshwater	*Potamogeton crispus* (curlyleaf pondweed) *Dreissena polymorpha* (zebra mussel) *Gambusia affinis* (western mosquitofish) *Myxobolus cerebralis* (whirling disease agent) *Phragmites australis* (common reed) *Pontederia crassipes* (water hyacinth)
14. Shelf ecosystems (neritic and intertidal/littoral zone)	Marine	*Sargassum muticum* (wire weed) *Carcinus maenas* (European shore crab) *Hemigrapsus sanguineus* (Asian shore crab) *Mytilus galloprovincialis* (Mediterranean mussel)
15. Open ocean pelagic systems (euphotic zone)	Marine	*Pterois volitans* (red lionfish) and *Pterois miles* (lionfish)
16. Deep sea	Marine	
17. Coastal areas intensively and multiply used by human	Marine	*Batillaria attramentaria* (Japanese false cerith) *Caulerpa racemosa* (green algae) *Caulerpa taxifolia* (killer algae) *Carcinus maenas* (European shore crab)

7. For more examples, see **Supplementary material 1.4**
8. Note that scientific names follow the taxonomy used in the original papers. Examples were chosen based on a systematic literature review. Data management report available at: https://doi.org/10.5281/zenodo.5518254

1.6.6 Nomenclature and taxonomy

The IPBES invasive alien species assessment generally follows the Global Biodiversity Information Facility (GBIF) Backbone taxonomy (GBIF, 2021), with a few exceptions for marine species, where authors have followed the World Register of Marine Species (WoRMS, 2022).

For increased accessibility where available, English common names, following the Centre for Agriculture and Bioscience International (CABI) Invasive Species Compendium (CABI, 2022) as the main reference source, are indicated alongside scientific names throughout the report.

The assessment acknowledges the diversity of common names across the globe, as well as their cultural importance (**section 1.6.7.1**). Common names are therefore sometimes included in the local language if pertinent to a specific case study, where such names are available and appropriate.

1.6.7 Cross-cutting themes

A number of cross-cutting themes have been acknowledged as important to IPBES assessments. In this assessment, three major cross-cutting themes are developed across chapters. 1) Indigenous Peoples and local communities are recognized as possessing detailed knowledge on biodiversity and ecosystems, and accordingly, IPBES is committed to promoting an enhanced recognition of and to working with Indigenous and local knowledge systems (Annex 1 of decision IPBES-7/1). 2) Good quality of life is included within the context of the IPBES conceptual framework and within the ongoing IPBES values assessment (IPBES, 2022a). 3) The Methodological Assessment of Scenarios and Models of Biodiversity and Ecosystem services (IPBES, 2016c) led to the commitment to continuing advanced work on scenarios and models of biodiversity and ecosystem functions. For each of the three cross-cutting themes, liaison groups were formed with representation of at least one expert from each of the chapters.

1.6.7.1 Indigenous and local knowledge

Engaging with Indigenous Peoples and local communities

Indigenous Peoples and local communities is a term used internationally by representatives, organizations, and conventions to refer to individuals and communities who either self-identify as Indigenous or as members of distinct local communities that maintain an inter-generational historical connection to place and nature through livelihoods, cultural identity, languages, worldviews, institutions, and ecological knowledge (IPBES, 2019). Indigenous Peoples and local communities are, typically, ethnic groups who are descended from and identify with the original inhabitants of a given region, in contrast to groups that have settled, occupied or colonized the area more recently (IPBES, 2019). At least a quarter of the global land area is traditionally owned, managed, used, or occupied by Indigenous Peoples, representing about 38 million km^2 (Garnett *et al.*, 2018). In addition, a diverse array of local communities, including farmers, fishers, herders, hunters, ranchers, and forest users, manage substantial areas under various property and access regimes (IPBES, 2019). Accordingly, Indigenous Peoples and local communities are stewards to an impressive diversity of nature's contributions to people (Brauman *et al.*, 2020; see **Chapter 2**, **Box 2.6**). However, these lands and waters may be increasingly impacted by invasive alien species (**Chapter 2**, **Box 2.6**; **Chapter 4**, **section 4.6**).

As a result of their close relationship with nature, and dynamic Indigenous and local knowledge systems, many Indigenous Peoples and local communities have developed new understandings and knowledge of biological invasions and invasive alien species (Howard, 2019; Jevon & Shackleton, 2015). They are observers to the introduction and spread of invasive alien species and their impacts on humans and biodiversity, often in environments where scientific monitoring (**Glossary**) and research are sparse or challenging. Many Indigenous Peoples and local communities have a good understanding of the often complex and interacting roles of drivers facilitating the introduction, establishment and spread of invasive alien species on their lands (**Chapter 3**, **Box 3.15**), and also employ their knowledge of the environment to develop responses or management strategies (**Chapter 5**) and are key, active participants in management and decision-making (**Chapter 6**, **section 6.4**; Fischer, 2007; Gratani *et al.*, 2011; Jagoret *et al.*, 2012). Indigenous Peoples and local communities are also in a position to judge trade-offs between beneficial and harmful impacts of invasive alien species both in terms of livelihoods and the environment, as they have to live with them or manage them in their lands and waters (S. J. Hall, 2009; Kannan *et al.*, 2016; Koichi *et al.*, 2012). For example, local authorities in Queensland, Australia, consulted with Girringun Aboriginal rangers and residents to better understand the extent of myrtle rust impacts on native plant species, and to design responses that align to the risk level posed, so as not to undermine local livelihoods (see also Grice *et al.*, 2012; Head & Atchison, 2015). Many Indigenous Peoples and local communities are therefore concerned that their knowledge, needs and views are not properly considered in both research and management of biological invasions (IPBES, 2020b, 2020b).

Working with Indigenous and local knowledge in the assessment

There is a clear need to work with Indigenous Peoples and local communities on assessments and activities related to biological invasions and invasive alien species. However, Indigenous and local knowledge is still often under-represented in research on biological invasion science, which represents a great loss to overall understanding and capacity to manage biological invasions and control invasive alien species. The IPBES invasive alien species assessment therefore aims to work with Indigenous and local knowledge, and to build its conclusions on the best available science and Indigenous and local knowledge. It recognizes that there are numerous barriers to effectively working with Indigenous and local knowledge in a global-scale assessment, including language, data and information flow, accessibility of information, representation of diverse groups within Indigenous communities, and differing understandings and conceptualizations of risk (e.g., Maclean *et al.*, 2021; Michán, 2011; Muller *et al.*, 2009). To overcome these issues as far as possible, the assessment follows the IPBES approach to recognizing and working with Indigenous and local knowledge (Decision IPBES-5/1, annex II), with the support of the IPBES task force and technical support unit on Indigenous and local knowledge. This work included convening three dedicated workshops[9] on Indigenous and local knowledge that brought together Indigenous Peoples and local communities and assessment authors (IPBES, 2020b, 2020b, 2022b), and the consideration of literature beyond the scientific journals

9. The first dialogue workshop took place in Montreal, Canada on 15-16 November 2019; the second dialogue workshop was held online from 21 September to 1 October 2020; and the third dialogue workshop was held online on 1-3 February 2022.

and major invasive alien species databases[10], including materials received through an online call for contributions for the assessment. Assessment authors also carried out an extensive cross-chapter review of literature on Indigenous and local knowledge. Consideration of free, prior and informed consent was key to this work.

The diversity of Indigenous Peoples and local communities' perspectives on invasive alien species

Indigenous Peoples and local communities' perspectives on invasive alien species often differ from scientific perspectives. Indigenous Peoples and local communities perceive invasive alien species in terms of both the particular ecological context and the cultural world views and traditions of their communities (Ellen, 2020). Science also brings its own set of value judgements relating to invasive alien species. This can lead to differences in understanding, responses, and management practices relating to biological invasions. Perspectives on any given invasive alien species will also vary within and between communities, as different community members may experience different impacts depending on gender, age, livelihood and a multitude of other factors (IPBES, 2022b). The great diversity of Indigenous Peoples and local communities' conceptions across species, places, cultures, livelihood systems and time periods, and consequential actions and responses to invasive alien species and the management of biological invasions, makes generalization almost impossible (IPBES, 2020b). Understanding these differing perceptions is therefore a key task for the assessment, and recognition of diverse perspectives is important if effective collaboration between scientists, policymakers, and Indigenous Peoples and local communities is to occur **(Box 1.13)**.

Many Indigenous Peoples and local communities emphasize the inter-relatedness of humans, the land, water, and other species (Barbour & Schlesinger, 2012), which can lead to acceptance of new species. For example, the Anishnaabe of the Great Lakes Region of North America explain the arrival of new plants or animals as a natural process of migration and must then determine why they have come and what their relationship with these migrants might be (Reo & Ogden, 2018). Thus, while some Anishnaabe support invasive alien species eradication, others argue: "…we're supposed to respect all of nature. To me having respect for nature is respecting the fact that it knows how to balance itself and stop trying to introduce different things to fix this and fix that…Respect nature and it will balance. I mean everything has its cycles, leave it alone for gosh sakes. Let it do its thing and quit playing God." (Reo & Ogden, 2018, quoting Kathy LeBlanc, a cultural leader and elder from the Bay Mills Indian Community).

In other cases, established invasive alien species have become a valued part of the socioecological system and are reflected in cosmology. Xeni Gwet'in and Tsilhqot'in of British Columbia now link their identity with *Equus caballus* (horses), describing them as relatives, individuals, or neighbours with family groups. As one elder put it, "The wild horses are like us. They've got routes they go to. They have plans… The mares are sort of the leaders, like in our culture the women have power. They are really respected and strong. So, the stud would protect the mares, but the mare would decide where to go, when to go. And it's quite interesting, in our culture it's the same" (Bhattacharyya & Slocombe, 2017).

Also, in some cases, the introduction of some invasive alien species occurred so long ago that these species can be perceived as native and now "belong to country" (Bach & Larson, 2017). Meanwhile, in many cases,[11] invasive alien species are perceived by Indigenous Peoples and local communities as "negative", often referred to as "weeds" or "pests", and "new" in contrast to "native species" often due to negative impacts on food systems, medicines, and livelihoods of Indigenous Peoples and local communities (IPBES, 2019b; **Chapter 4**, **section 4.6**). A further key issue can be that Indigenous Peoples and local communities' cosmologies or cultural world views may not have a place for these new species: invasive alien species may often be seen as a cultural and spiritual threat, as well as an ecological issue (Grenz, 2020; IPBES, 2020b, 2022b; Trauernicht *et al.*, 2013). For example, among the Māori of New Zealand, Peltzer *et al.* (2019) report that introduced predators have significantly challenged the key cultural concept of "whakapapa", which portrays the genealogical connections between the natural world, including humans, and the cosmological domain. Similarly, among some Australian Aboriginal groups, invasive alien species are a threat because they have no dreaming – no origins accounted for in the ancestral creation of the landscape – and thus no law or responsibilities assigned to families to care for and respect them (Crowley, 2014; Salmón, 2000). Some Indigenous Peoples and local communities explain dramatic and especially negative changes in the landscape, such as an invasive alien species, as a failure of humans to uphold their responsibilities: For example, the Soliga describe the establishment and spread of the invasive alien plant *Lantana camara* (lantana) in Southern India as the punishment of the Hindu Lord Shani for unknown moral infringements by the local communities (Puri, 2015; Thornton *et al.*, 2019).

10. Data management report available at: https://doi.org/10.5281/zenodo.5760266

11. Data management report available at: https://doi.org/10.5281/zenodo.5760266

Box 1 13 **Indigenous and local knowledge of invasive alien species in names, stories, and songs.**

Indigenous and local knowledge of invasive alien species may be embedded in stories, poetry, and songs. A poem from Ethiopia illustrates local understandings of the adverse impacts of invading *Prosopis juliflora* (mesquite, or woyane harar trees) on fodder resources and cattle grazing practices, and their interactions with other drivers of change in nature:

> "Cattle from upland, cattle from lowland
> Goats from here, sheep from there
> Are you [my camels] ever going to have the trees
> That you once had all for yourselves?
> In the summer, the floods
> In the winter the locusts
> In the upland the Christians
> On the lowland the sorghum fields
> In awash the woyane trees
> Where should I take you my heart [my she camel]?"
> (Balehegn, 2016)

Indigenous and local knowledge of biological invasions may also be embedded in specific names, which may also reveal much about how an invasive alien species is perceived. Most invasive alien species are given new names by Indigenous Peoples and local communities, which may indicate origin or foreignness as well as inclusion in a similar generic category, and can have political undertones. For example, the Kawaiwete of Brazil label the incoming, and more aggressive, hybrid African-European honey bee as a "honey wasp", in contrast to the benign local "honey bee" (Athayde *et al.*, 2016). In Kenya, the introduction of the invasive alien tree *Prosopis juliflora* is locally dubbed woyane harar after the Tigrinean People's Liberation Front (TPLF), which introduced the tree for land reclamation, fodder, and wood fuel (Berhanu & Tesfaye, 2006; Rettberg, 2010; Tessema, 2012). *Chromolaena odorata* (Siam weed) is known as rumput golkar or golkar grass in Timor after the ruling government party of Indonesia, as it overshadows competitive plants (McWilliam, 2000). Similarly, Congress grass refers to the poisonous *Parthenium hysterophorus* (parthenium weed) across India, said to have been inadvertently gifted to the nation in wheat that was imported for famine relief by Nehru's Congress Party in the mid-1950s (OpIndia, 2021). Invasive alien salmonids in the fresh waters of Argentinian Patagonia are known to the Mapuche as cosa de winka ("white man stuff"), associated with the arrival of settlers who introduced these environmentally damaging species for sport fishing; they are now considered as ill omens that disturb native fish populations and the sacred status of the waters and their inhabitants (Aigo & Ladio, 2016). More positively, *Prosopis juliflora* is welcomed by many in Jordan, despite acknowledging its negative impacts, as a source of vegetation cover, fodder, firewood, and charcoal, and is known as Al salam ("the peace"; Al-Assaf *et al.*, 2020).

As noted above, diversity in perception may also occur within communities. In Chitwan National Park, Nepal, Tharu household socioeconomic characteristics influence the perceived value of invasive alien *Mikania micrantha* (bitter vine). Those families that were more dependent on forest products incurred more of both the costs and the benefits associated with *Mikania micrantha* than less forest dependent families (Murphy *et al.*, 2013; Rai & Scarborough, 2015). *Sus scrofa* (feral pig) in Northern Australia is similarly either vilified for its negative impacts on vegetation, soils, other wild foods, cultural heritage sites, and because it increases the spread of invasive alien *Lantana camara* (lantana), or highly valued as an important food source for those with lower socioeconomic status (Koichi *et al.*, 2012). Likewise, there are diverging perspectives on *Bubalus bubalis* (Asian water buffalo) and *Equus caballus* (horses) in Northern Australia, with many worried about damage to sacred sites and wild foods, while others benefit from them directly or want financial returns from animals when they are controlled (Ens *et al.*, 2016; **Chapter 4**, **Box 4.14**). Underemployed or low income Māori have benefited from invasive alien species, such as products from possums and pacific rats, while other Māori see them as both ecological and cultural threats (Peltzer *et al.*, 2019). Similarly, Hawaiian cattle (Fischer, 2007), and *Camelus dromedarius* (camels) and *Bubalus bubalis* in Australia (Vaarzon-Morel, 2010; Weston *et al.*, 2012) have been viewed in mixed fashion. Overall, the different perceptions within and between communities, caused by gender, age, knowledge status, livelihoods, and spirituality, result in a diversity of viewpoints on management and policy options for biological invasions (**Chapters 5** and **6**).

1.6.7.2 Good quality of life

Invasive alien species not only affect biodiversity and the ecological processes underpinning nature's contributions to people, but they also directly or indirectly affect good quality of life (or human well-being). Good quality of life is the achievement of a fulfilled human life, a notion which varies strongly across different societies and groups within societies. It is a context-dependent state of individuals and human groups, comprising access to food, water, energy and livelihood security; health, good social relationships; equity, security, cultural identity; and freedom of choice and action **(Table 1.4)**. Much of this provision is a result of nature's contributions to people **(Figure 1.12**; **Box 1.12)**, but its fair distribution and progressive attainment relies principally on governance arrangements and social capital/infrastructure. Good quality of life and health encompass not just physical health, but psychological health, including the satisfaction created by cultural expression and stability, spiritual fulfilment, and reliable access to the resources necessary to thrive as a human being. Though people generally introduce alien species deliberately in order to improve their incomes, food security, or tangible material assets, invasive alien species can threaten good quality of life in various ways at both the individual and community level **(Box 1.9)**; but it can also be argued that efforts to manage invasive alien species can be seen in some cases as detrimental to good quality of life, especially if they involve the cessation of access to natural resources for some groups in society, or inappropriate use of hazardous chemicals. There are also clear cases where communities have adapted to invasive alien species (**Chapter 6**, **section 6.2.2.5**), sometimes because they lacked other options (IPBES, 2022a) and where this has enhanced local good quality of life. Although the preponderance of evidence suggests that invasive alien species are mainly viewed as threats and challenges to human communities, at least one recent study indicates that adaptation is a more dominant response than eradication efforts (Howard, 2019).

It follows that management techniques and policy development will likely benefit from taking into careful consideration the trade-offs among different constituents of good quality of life. For example, people might be willing to accept reductions in their resources, safety, health or lifestyle choices for what they consider a greater cause, such as community survival or national pride. Furthermore, communities will not necessarily be united in how they feel about the values of invasive alien species and associated detrimental or beneficial impacts (Kelsch *et al.*, 2020; Shackleton, Larson, *et al.*, 2019). Many citizens may feel quite neutral or are apathetic about the issue.

There is also the question of scale. While it is obvious that good quality of life encompasses individuals and small communities, it can also refer to national or even supranational identities, stability, survival and resilience. For example, framing invasive alien species as a local problem as opposed to a national security issue will have an impact on policy response options and levels of related resource allocation (Stoett, 2010). Ultimately, considering good quality of life across scales and linking levels of governance will improve the management of biological invasions (**Chapter 6**, **section 6.3.1.1**). **Table 1.4** below presents some examples of constituents of good quality of life which have been considered in the present assessment.

Another prominent element affecting good quality of life is the differentiation in status and access to resources related

Table 1 4 **Constituents of good quality of life and examples of their subcategories.**

The overarching premise for all constituents is the freedom of choice and action, that is, the opportunity to be able to achieve what a person values doing and being. Adapted from Bacher *et al.* (2018); Millennium Ecosystem Assessment (2005).

Constituents of human well-being	Examples
Safety – human security	Personal safety Gender equality Secure resource access Security from disasters Resilient communities
Material and non-material assets	Adequate livelihoods Sufficient nutritious food Shelter Access to goods Recreation
Health	Physical health Feeling well/psychological health Access to clean air and water Absence of infectious disease
Social, spiritual and cultural relations	Social, spiritual and cultural practice Social infrastructure and governance Environmental, social justice and equity Mutual respect Friendship Identity and autonomy
Freedom of choice and action	Control over events and actions

to gender. There is limited research on the interplay between gender relations and invasive alien species, but it is clear that women can be impacted differently in cases where they are expected to engage in many of the forms of labour that are most directly affected by invasive alien species, such as health care, firewood gathering, and the acquisition and use of water for cleaning, sanitation, or family consumption (Fish *et al.*, 2010; Shrestha, 2021). Women are often tasked with the difficult (and often futile) job of weeding by hand, which can take up valuable time better spent on other quality-of-life-related tasks and expose them to dangerous pesticides and herbicides (Terefe *et al.*, 2020). The sharp thorns of the invasive alien *Prosopis juliflora* (mesquite) shrub (native to Mexico, introduced in Ethiopia in 1999) harm the hands of women collecting fuel wood (Terefe *et al.*, 2020). It has also been suggested that personal safety can be compromised with the advent of invasive alien plants; for example, local reports of sexual assault under cover of dense stands of invasive alien *Acacia* spp. invasions have been made (Shackleton, Shackleton, *et al.*, 2019; de Neergaard *et al.*, 2005). More international research on the role of gender in invasive alien species identification, management, and monitoring is needed for a more nuanced perspective to emerge.

The succession of emerging zoonotic diseases in the early twenty-first century has led to the development of several holistic and interdisciplinary approaches to safeguard health. Current concepts such as Planetary Health, EcoHealth, and One Health (**Glossary**) stress the importance of understanding the links between human, animal, and environmental health, though with a strong emphasis on safeguarding the health of vertebrates (Lerner & Berg, 2017). The World Health Organization (WHO), the Food and Agriculture Organization of the United Nations (FAO), and the World Organisation for Animal Health (WOAH, founded as OIE) provide international standards for human health, plant health, and animal health, respectively. Working together with the UNEP through a One Health High-Level Expert Panel (OHHLEP), they have jointly defined One Health as "an integrated, unifying approach that aims to sustainably balance and optimize the health of people, animals and ecosystems. This approach recognizes that the health of humans, domestic and wild animals, plants, and the wider environment (including ecosystems) are closely linked and inter-dependent. It mobilizes multiple sectors, disciplines and communities at varying levels of society to work together to foster well-being and tackle threats

Box 1 14 The role of invasive alien species in zoonotic disease transmission.

The relationship between invasive alien species and human health, particularly pathogenic microbes, and emerging infectious diseases (Pyšek, Hulme, *et al.*, 2020) is especially relevant in a decade which began with a global coronavirus disease (COVID-19) pandemic that killed close to 20 million people (The Economist, 2022), ravaged the world economy, and exacerbated inequality and poverty (Ritchie *et al.*, 2020). Invasive alien species can have serious implications for human health (Lazzaro *et al.*, 2018; Pyšek & Richardson, 2010): alien species can act as a vector of pathogens (e.g., *Aedes albopictus* (Asian tiger mosquito) for dengue fever; Brady & Hay, 2020; Hulme, 2014); produce allergenic pollen (*Ambrosia artemisiifolia* (common ragweed); Richter *et al.*, 2013); and be poisonous (e.g., *Rhinella marina* (cane toad); Bacher *et al.*, 2018) or venomous (e.g., sea jellies; Kideys & Gücü, 1995). Indeed, the COVID-19 pandemic has demonstrated the catastrophic consequences of ongoing environmental transformation, wildlife exploitation, and the movement of organisms in a globalized world (IPBES, 2020c; Nuñez *et al.*, 2020).

Parasites (including pathogenic bacteria, fungi and viruses) can be introduced into an invaded range alongside an invasive alien species (Bojko *et al.*, 2021; Dasgupta, 2021; Daszak *et al.*, 2000; Evans, 2003; Roy *et al.*, 2017). Additionally both introduced and endemic parasites can change the strength of interactions between species and ultimately affect the outcome of a biological invasion (Amsellem *et al.*, 2017; Dunn & Hatcher, 2015). Pathogens causing emerging infectious diseases (WHO, 2014), which spread into new host populations or species, are rarely treated as invasive alien species, but it is widely recognized that the introduction of novel organisms (those without evolutionary analogues in the recipient environment) have the potential to be incredibly disruptive (Nuñez *et al.*, 2020; Saul & Jeschke, 2015; Vilà *et al.*, 2021).

The role of invasive alien species in the transmission dynamics of emerging zoonotic diseases is often overlooked (Nuñez *et al.*, 2020; Vilà *et al.*, 2021) despite the interlinkages between human health and biodiversity loss having now been explored in great detail by the scientific community (Estrada-Peña *et al.*, 2014; Jones *et al.*, 2008; UNEP *et al.*, 2015; Wolfe *et al.*, 2007). Integrated approaches that take into account the landscapes and seascapes in which socio-ecological systems, including their human dimensions, are embedded, could be part of an effective collective response to the threats posed by invasive alien species and related pathogenic diseases. Invasive alien species are part of these broader systems, and the harm to human health which results from their spread and from emerging infectious diseases share many characteristics **(Figure 1.17)**. Pathogenic microbes which cause human epidemics and pandemics are highly successful invasive alien species, transmitted by human behaviour. Integrated interdisciplinary approaches will contribute to increased understanding of the interplay amongst factors driving disease transmission whether in humans, other animals or plants (Vilà *et al.*, 2021).

to health and ecosystems, while addressing the collective need for clean water, energy and air, safe and nutritious food, taking action on climate change, and contributing to sustainable development" (UNEP, 2021). There have been previous efforts to integrate biological invasions within the One Health approach (Conn, 2014; Hulme, 2020a; F. A. B. Meyerson *et al.*, 2009; L. A. Meyerson *et al.*, 2002; L. A. Meyerson & Reaser, 2002, 2003). However, despite the critical role invasive alien species can play as reservoirs and vectors of zoonotic diseases (**Box 1.14**; Hulme, 2014; Roy *et al.*, 2017, 2023), Planetary Health, EcoHealth, and One Health approaches have yet to systematically integrate the threat and impacts of biological invasions into their analyses (IPBES, 2020c; Chinchio *et al.*, 2020; Bertelsmeier & Ollier, 2020; Nuñez *et al.*, 2020; Vilà *et al.*, 2021). The acceptance of the One Health approach as appropriate by many governments and international organizations might change this, however. A more biosecurity-focused approach has also been suggested: "One Biosecurity" would integrate the One Health framework with the practical necessities associated with the provision of biosecurity, including the prevention of all invasive alien species (Hulme, 2020b; **Glossary**). One Biosecurity could be informed through a streamlined approach to the prediction of emerging biosecurity risks (whether pathogens, pests, or weeds), a global network of surveillance (**Glossary**) and information sharing, and coordinated international responses to incursions of invasive alien species. Such an approach could be underpinned by a regulatory framework that parallels the International Health Regulations of the WHO (Hulme, 2021) (**Chapter 6**, **section 6.7.2.2**).

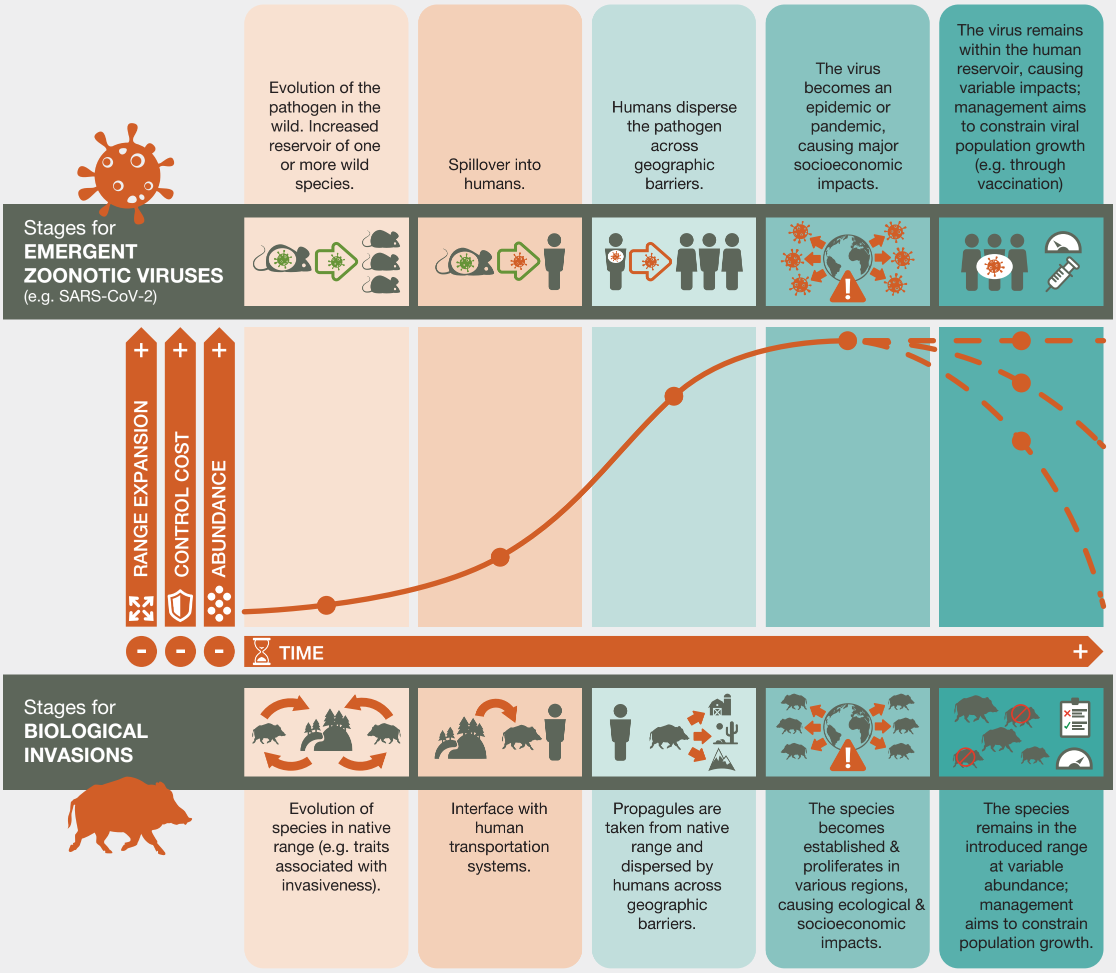


Figure 1 17 **Comparison of the stages of a zoonotic viral epidemic and those of a biological invasion.**

Source: Nuñez *et al.* (2020), https://doi.org/10.1016/j.tree.2020.05.004, under license CC BY 4.0.

1.6.7.3 Scenarios and modelling

Understanding the drivers and patterns of invasive alien species dynamics is crucial for designing and implementing appropriate management and monitoring strategies (Brundu & Richardson, 2016). There is a growing need to reconstruct the routes of introduction of invasive alien species (Estoup & Guillemaud, 2010; Gautier *et al.*, 2022) to predict biological invasions and effectively support different types of intervention, from early detection to management of established invasive alien species (S. A. Hall *et al.*, 2021; Van Wilgen *et al.*, 2011). Indeed, the importance of model- and scenario-based prevention and early detection has been highlighted in several policies including the European Union Regulation 1143/2014 on invasive alien species (European Union, 2014). Modelling approaches have been used to define coarse climatic envelopes for invasive alien species (Brundu & Richardson, 2016; Pino *et al.*, 2005), and reconstructing routes of biological invasions (Gautier *et al.*, 2022). Fine-scale species distribution modelling and prediction requires information on local environmental and habitat factors (Vicente *et al.*, 2011), as well as linking correlative models to demographic variables or demography-based population models (Kueffer *et al.*, 2013; Vicente *et al.*, 2019). The prevention, early detection and management of biological invasions will consequently benefit from increased knowledge, more informative predictions, and accurate and plausible future scenarios (Chornesky *et al.*, 2005; Genovesi & Monaco, 2013; Roura-Pascual *et al.*, 2021).

For invasive alien species, scenarios and models have been applied to inform understanding of how spatial-temporal patterns emerge (**Chapter 2**, **section 2.6.5**; **Chapter 4**, **section 4.7.1**), of which processes are underlying these patterns, and of how ecological, economic, and societal drivers relate to the emergence of the observed patterns (**Chapter 3**, **Box 3.14**). Scenarios and models differ in their approach to investigate historic, current, and future patterns of alien species richness, abundance and distributions. While models aim to predict alien species patterns based on how environmental, economic or social variables relate to species occurrence or abundance, scenarios are based on alternative possible future states of those variables resulting in projections of potential future patterns of biological invasions (IPBES, 2016c; Lenzner *et al.*, 2019; Roura-Pascual *et al.*, 2021). In the section below, scenarios and models are briefly contrasted in terms of how patterns and dynamics are analysed, the methods used, their different uses, and the advantages and disadvantages of each approach. A systematic review was undertaken to assess the current use of scenarios and models within the context of biological invasions.[12]

12. Data management report available at https://doi.org/10.5281/zenodo.5706520

Models

Models can be defined as “qualitative or quantitative representations of key components of a system and of relationships between these components”.[13] There are four broad groups of model types (main model types) identified (IPBES, 2016c):

i. Expert-based models include any type of qualitative expert opinion (where experts are defined as a single person or group of people that hold specific knowledge of a process, species or system of interest). Experts may include scientists and other academics, relevant stakeholders and Indigenous Peoples and local communities (**section 1.6.7.1**).

ii. Correlative models (also called statistical models) use empirical data to estimate parameter values for processes that are implicit rather than explicit.

iii. Process-based models (also mechanistic models) explicitly integrate processes or mechanisms based on established scientific understanding.

iv. Hybrid models combine correlative and process-based modelling approaches.

Most papers identified through the systematic review used correlative models (57 per cent of 781 observations), followed by process-based models (33 per cent), hybrid models (8 per cent) and expert-based systems (1 per cent).

There are also interdisciplinary models and integrated assessment models (IPBES glossary[12]) that are used to describe the complex relationships between environmental, social and economic drivers (e.g., Havlík *et al.*, 2014) by integrating trans-disciplinary knowledge to capture large-scale dynamics, interactions and feedbacks of a specific system (Harfoot *et al.*, 2014). Integrated assessment models assess “wicked problems” which are highly complex, socioecological problems including many variables and actors (Termeer *et al.*, 2019). Currently, biological invasions are not included in existing global integrated assessment models, but such an integration would be highly beneficial (Lenzner *et al.*, 2019).

Further details, including opportunities and limitations, of these modelling approaches are provided in the data management report.

Scenarios

Scenarios are “representations of possible futures for one or more components of a system, particularly for drivers of change in nature and nature’s benefits, including

13. IPBES glossary: https://ipbes.net/glossary

alternative policy or management options".[14] Different types of scenarios can be identified and are applicable in specific contexts:

i. (Exploratory scenarios (also called "explorative scenarios" or "descriptive scenarios") examine a range of plausible futures, based on pre-defined drivers and their assumed future trajectories starting from the present conditions.

ii. Target-seeking scenarios (also called "goal-seeking scenarios" or "normative scenarios") have a clear objective or set of objectives for a point in time in the future (i.e., a specific target) and aim to describe plausible pathways to achieving this outcome. The procedure of developing such scenarios is called backcasting.

iii. Policy-screening scenarios aim to evaluate alternative policy or management options. They either follow a similar logic to target-seeking scenarios where a future policy goal is determined, or they can be developed through policy screenings (also called "ex-ante scenarios"). See the IPBES glossary and the methodological assessment report on scenarios and models of biodiversity and ecosystem services (IPBES, 2016c) for more detail.

Most of the papers identified through the systematic review focused on exploratory scenarios (87 per cent of papers), followed by policy-screening (7 per cent) and target-seeking scenarios (6 per cent). In most papers, scenarios were quantitative (82 per cent) as opposed to qualitative (9 per cent) or both quantitative and qualitative scenarios (8 per cent).

Overall, scenarios aim to provide a holistic view on global trends and processes and how they might shape the world's future under different assumptions. For many drivers of change in nature (e.g., climate; IPCC, 2014) and socioeconomic domains (e.g., demography, land-use; Hurtt *et al.*, 2011), such scenarios have already been developed. However, biological invasion scenarios have not been available until recently (Corrales *et al.*, 2018; Dehnen-Schmutz *et al.*, 2018; Ricciardi *et al.*, 2017). The need for scenarios for short (2030), mid (2030-2050), and long-term (2050-2100) trends in alien species richness and distribution at various scales to inform targets has been recognized (Bellard *et al.*, 2013; Roura-Pascual *et al.*, 2021). Increasing data availability and increased understanding of (historic) trends, distribution and impacts of invasive alien species globally and locally makes the development of scenarios for biological invasions feasible (Lenzner *et al.*, 2019). Recently, the first alternative futures for biological invasions were published (Roura-Pascual *et al.*, 2021). Roura-Pascual and colleagues developed 16 different qualitative scenarios storylines, which can be grouped into four archetypes based on their description of potential futures. The scenarios develop potential future trajectories of the world until 2050 with a special focus on drivers relevant for biological invasions (Essl *et al.*, 2020) and projected changes in alien species richness.

Moreover, recently IPBES has developed a framework for the creation of independent multiscale biodiversity scenarios for constructing pathways towards desirable futures for nature – the Nature Futures Framework (IPBES, 2022d). A distinguishing feature of the Nature Futures Framework, beyond classical environmental scenario frameworks, is the consideration of a plurality of perspectives and values towards nature within the scenarios, facilitating the assessment of different views on nature and ensuring the integration of these views through participatory approaches. While the Nature Futures Framework has not yet been applied in the context of biological invasions, it has considerable potential for exploring the role of invasive alien species in future biodiversity change across scales and contexts.

Scenarios and models in invasive alien species research

The scenarios and models' liaison group undertook a systematic review[15] including an initial set of 30,299 research papers of which 778 research papers were found to consider both the use of models and scenarios to evaluate the patterns and trends of invasive alien species. The search was restricted to indexed publications in English, ensuring a structured, systematic approach to the use of the terms "invasive alien species", "modelling" and "scenarios". A summary of the outcomes is provided here with further information available in the data management report.14 In some cases, a single paper focused on multiple categories (e.g., a model applied to both the United Kingdom and Portugal), and these are categorized as separate observations. The information is summarized as either a percentage of papers or of observations.

Patterns and trends

The Americas was the IPBES region with the highest proportion of observations across all papers, with 33 per cent of all observations (total number of observations: 1,153), followed by Europe and Central Asia (26 per cent), Asia and the Pacific (24 per cent), Africa (13 per cent) and finally Antarctica (2 per cent). In 3 per cent of the papers, the IPBES region was not stated. Most papers focused on only

14. IPBES glossary: https://ipbes.net/glossary

15. Data management report, including full output of the review, available at https://doi.org/10.5281/zenodo.5706520

one IPBES region (78 per cent of a total of 778 papers) and one country (70 per cent).

Most of the papers (63 per cent of all papers) were focused on only one invasive alien species with most focusing on invasive alien plants (including bryophytes; 40 per cent of observations from a total of 858 observations), followed by invertebrates (30 per cent), fishes (8 per cent), mammals (7 per cent), amphibians, birds and reptiles (3 per cent); and finally, fungi (2 per cent) or other invasive alien species taxa such as algae, bacteria, virus or protozoan (2 per cent). Furthermore, the majority of papers focused on only one particular IPBES unit of analysis (96 per cent of 778 papers), with the terrestrial environment dominating the literature extracted from the review with 75 per cent of observations (from a total of 813), followed by the freshwater (15 per cent) and the marine (8 per cent) environments. The impacts of invasive alien species were addressed in only 22 per cent of papers with most of these papers focusing on negative impacts (18 per cent of all papers). Invasive alien species pathways were considered in only 10 per cent of papers. Only 23 per cent of papers (n=182) considered invasive alien species management, and most papers focused on one (54 per cent) or two management strategies in combination (37 per cent).

The cross-cutting themes identified for the IPBES invasive alien species assessment were poorly represented in the papers with only 1 per cent considering Indigenous and local knowledge, 3 per cent considering good quality of life and 6 per cent including nature's contributions to people.

Further descriptive summaries and results from the review, including multidimensional scaling, illustrating the clustering of model and scenario features from across the papers, are available in the data management report.[16] Further specific detailed information from the review is included within the relevant chapters.

1.6.8 Key issues in the discussion of biological invasions

Throughout this assessment several key issues, some extant and some emerging, have been identified as critical to the discussion of biological invasions. The key issues identified within this assessment include the advent of **globalization**, the **impact of global environmental change** (and, in particular, the global biodiversity crisis), the **use of adaptation strategies**, the **role played by technology**, the **challenges for islands and protected areas**, and the **role micro-organisms play** in the broader understanding of invasive alien species.

16. Data management report available at https://doi.org/10.5281/zenodo.5706520

The most obvious issue is that of **globalization**, which has acted as an important overarching driver facilitating the unprecedented spread of invasive alien species that humans face today. There is a strong historical link between colonization by European powers and biological invasions, and the rise of global transport and trade has been a primary driver responsible not only for the transport and introduction of invasive alien species but also for the advent of biotic homogenization, which lowers resilience and increases vulnerability to further invasive alien species. Globalization is a catalyst exacerbating the problems of a human-dominated biosphere that has led to the Anthropocene, a world with biophysical systems profoundly shaped by human activity. The increasing levels of invasive alien species on a global scale are stark evidence of this era. At the same time, international instruments developed to prevent the spread of invasive alien species rely heavily on international organizations that are at least partially reflective of the process of globalization.

Another central key issue is the present and future **impact of global environmental change**, and the underlying direct and indirect anthropogenic drivers of change, not only on the spread and introduction success of invasive alien species but also on options for management (**Chapters 3** and **5**). Climate change and land and sea use, but also pollution (chemical, plastics, debris, etc.), ocean acidification, and other systems-level direct drivers of change in nature are currently shaping the Anthropocene, and driving, in particular, the loss of biodiversity (IPBES, 2019). Invasive alien species have long been identified as one of the primary drivers of this global biodiversity crisis, and they interact with other drivers of global environmental change to exacerbate it (**Chapters 3** and **4**).

The overarching issue of **human community adaptation** is noticeable as well: While invasive alien species can cause both harm and benefits, some human communities (at various scales, from rural areas to Indigenous Peoples and local communities to cities to regions) have in fact adapted to the presence of invasive alien species, and it is informative to see how, why, and in what forms this adaptation took place over time. This key issue, which is even more pertinent in the current era where climate change is forcing unprecedented adaptation and evolving survival strategies, is discussed more explicitly in **Chapter 6**, **section 6.2.2.5**. In some cases, the response to invasive alien species does not adequately deal with the threats they pose, and adaptation may be the only or the preferred policy response. It is important to note that prevention is an effective approach to managing invasive alien species and the costs of responding to biological invasions far outweigh the costs of prevention (Diagne *et al.*, 2021). However, in some cases, invasive alien species have become part of socio-ecological systems and are here to stay.

The evolving **role played by technology** is another key issue. The development of the steam engine enabled faster trans-ocean voyages involving ballast water usage, thus acting as a driver that accelerated pathways for

Box 1 15 The role of citizen (or community) science in monitoring invasive alien species.

Citizen science (also known as community science, participatory monitoring, community-based environmental monitoring, crowd science, crowd-sourced science, civic science, or volunteer monitoring) is a term that describes the diverse range of approaches in which scientific research is conducted, in whole or in part, by volunteers with varying levels of expertise (Gura, 2013; Pocock *et al.*, 2014, 2018). Citizen science is defined by the European Commission Green Paper as "general public engagement in scientific research activities where citizens actively contribute to science either with their intellectual effort, or surrounding knowledge, or their tools and resources" (Consortium, 2013; Follett & Strezov, 2015).

People contribute to biodiversity and ecosystem research through citizen science in diverse ways including providing data, raising new research questions, and communicating and disseminating findings. Citizen science can be broadly considered as contributory or collaborative (co-created). Within contributory citizen science, participants are primarily involved in data collection while through collaborative citizen science, participants are involved in various stages of the scientific process including identifying the scope and research questions through to interpreting and using the results. Citizen science not only results in scientific advances but is also known to increase public understanding of science by improving the scientific capacity of participants through skills acquisition and learning (MacPhail & Colla, 2020; Steven *et al.*, 2019).

There are many diverse approaches to surveillance and monitoring of invasive alien species. Citizen science is seen as particularly relevant for environmental monitoring and has a long history in many countries with some initiatives in Northern Europe and North America having been ongoing for more than a century (Allen, 1976; Miller-Rushing *et al.*, 2012; Pocock *et al.*, 2015). Many of the large-scale and long-term global biodiversity datasets have relied on contributions from volunteers. Indeed, citizen science is often used to engage people in scientific projects that may be impractical for individuals or small groups to conduct alone because of the need to gather or analyse "big data" (Willett *et al.*, 2013).

Volunteers have made substantial contributions to understanding biological invasions (Roy *et al.*, 2015) from documenting the arrival, establishment, and spread of alien species through to predicting potential new arrivals through horizon scanning (Roy *et al.*, 2020) and so contributing to early-warning. The breadth of expertise provided by taxonomic experts from volunteer biological recording communities is essential for horizon scanning. Prioritization of invasive alien species through horizon scanning can be used to inform mass participation approaches involving the public (or where relevant special interest groups such as anglers) in monitoring and surveillance underpinning early-warning.

The advent of mobile computing technologies in smartphones and tablets and the corresponding proliferation of mobile applications (apps) have greatly expanded the potential of citizen science for contributing to research on invasive alien species (Adriaens *et al.*, 2015). As mobile phones become increasingly ubiquitous (users now exceed 2.8 billion people worldwide; Alavi & Buttlar, 2019), citizen science is undergoing an unprecedented shift in the scale and quantity of available data (Silvertown, 2009; Teacher *et al.*, 2013). Popular biodiversity reporting apps like eBird (Sullivan *et al.*, 2014) and iNaturalist (Unger *et al.*, 2021) have user communities in the hundreds of thousands, generating enormous quantities of data for research (e.g., over 1 million records in iNaturalist in the first seven years; Pimm *et al.*, 2014). Invasive alien species reporting apps, which enable users to submit geotagged observations of invasive alien species, are an excellent new source of spatiotemporally explicit occurrence data for invasive alien species management and research, and seen as a major pathway to implementing surveillance and monitoring at national and global scales (Martinez *et al.*, 2020). The number of invasive alien species reporting apps available is steadily increasing, ranging from regional apps to those focused on particular taxa including aquatic organisms, insects, and plants (e.g., Goëau *et al.*, 2013; Laforest & Bargeron, 2011; Scanlon *et al.*, 2014; Wallace *et al.*, 2020).

Many mobile devices now include a variety of onboard sensors and instrumentation like barometers, gyroscopes, accelerometers, microphones, cameras and ambient light sensors, and the capability of storing data from these sensors and uploading it to online databases (Lane *et al.*, 2010). Onboard sensors are increasingly used to facilitate and even automate citizen science participation *via* invasive alien species apps, for example in bioacoustics surveys for invasive alien amphibians (Platenberg *et al.*, 2020). Artificial intelligence and machine learning, especially in image recognition, are further enhancing mobile app contributions to citizen science, by allowing for the automated identification of organisms in user-submitted images (Terry *et al.*, 2020). The steady improvement and increasing availability of online invasive alien species occurrence databases and their integration with mobile technology is another major and ongoing advance underpinning citizen science (Martinez *et al.*, 2020; Reaser *et al.*, 2020; Seebens *et al.*, 2020).

Science-society-policy interactions are developed through open and collaborative approaches amongst participants involved in citizen science (Powell & Colin, 2009; Gardiner & Roy, 2022). Collaborative research outcomes, resulting from open, networked and transdisciplinary citizen science approaches, can ultimately contribute to democratic decision-making.

the transport of invasive alien species (sailing ships also needed ballast but used soil, which itself carried invasive alien species but at slower delivery times) (**Chapter 3**, **section 3.2.3**). Modern technology (including genetics/genomics, informatics, and drone surveillance) is facilitating the transport of alien species around the globe *via* e-trade (**Chapter 3**, **sections 3.2.3** and **3.2.4**), but are also being used in new and inventive ways to discover, track, and manage invasive alien species and their impacts (**Chapter 5**). New online tools and technologies, particularly new data streams and data integration methods, will increase capacity to deliver a global monitoring and decision-support system for managing biological invasions (Martinez *et al.*, 2020; McGeoch & Jetz, 2019). Relatedly, communication strategies in the internet age have emerged as fundamental as people share new information about identifying and dealing with invasive alien species. Citizen science (**Glossary**), including approaches that encompass visual identification technologies and other innovations, has become a popular and valuable approach to underpin research and policy on biological invasions and invasive alien species (**Box 1.15**; Encarnação *et al.*, 2021; Roy *et al.*, 2015).

Insular environments, from oceanic islands and deep sea hydrothermal vents to freshwater systems and fragmented habitats, have provided insights into the relationships between geographic patterns and biological processes (D. R. Drake *et al.*, 2002). Such insular systems feature prominently in this assessment. Islands, especially SIDS, are considered particularly vulnerable to invasive alien species because of the difficulty of prevention where globalization, including mass tourism, has become deeply integrated into island economies. Invasive alien species on islands have been shown to have some of the most detrimental impacts compared to continental ecosystems, including the extinction of many endemic species (e.g., Bellard *et al.*, 2016; Pyšek, Blackburn, *et al.*, 2017). Indeed, invasive alien species are ranked as the leading cause of biodiversity loss on islands (Bellard *et al.*, 2016; Russell *et al.*, 2017). However, there are many examples whereby management of invasive alien species, including approaches to prevent arrival and eradication of specific taxa, has proven successful on islands (**Chapter 5**; Courchamp *et al.*, 2003; Russell *et al.*, 2017).

While most invasive alien species tend to thrive in anthropogenically disturbed ecosystems, some species are able to reach even the most remote and well conserved areas, including those formally declared as **protected areas** (Liu *et al.*, 2020; **Chapter 4**, **section 4.3.1.2**). Indeed, it is clear that the establishment of protected areas, in both terrestrial and marine environments, does not preclude the unintentional introduction and spread of invasive alien species, such as those associated with illegal wildlife trade and other activities such as fishing and recreation without high biosecurity standards. Indeed, there are concerns that biological invasions are insufficiently considered when devising management plans for marine protected areas in particular (Galil, 2017; Giakoumi *et al.*, 2016). Furthermore, historic or current intentional introductions such as through afforestation projects associated with climate change mitigation efforts can pose a threat to protected areas worldwide (Richardson, 1998), and ecosystem restoration projects also face similar concerns.

Another important key issue within biological invasions is the consideration of **microorganisms**, from virus to protozoa, including the links between invasive alien species and plant, animal, and human diseases including zoonotic diseases such as COVID-19, H1N1 flu (swine flu) and viral haemorrhagic fever (Ebola; **Box 1.14**). Such microorganisms have profound implications for good quality of life (Amsellem *et al.*, 2017) and biosecurity (Hulme, 2020a), and create space for further discussions of ecosystem-based and One Health approaches.

These key issues are relevant to natural science, social science, the humanities and policy developments, and will likely shape the evolution of our understanding of the biological invasion process in the years to come.

REFERENCES

Abramides, G. C., Roiz, D., Guitart, R., Quintana, S., Guerrero, I., & Giménez, N. (2011). Effectiveness of a multiple intervention strategy for the control of the tiger mosquito (*Aedes albopictus*) in Spain. *Transactions of the Royal Society of Tropical Medicine and Hygiene, 105*(5), 281–288. https://doi.org/10.1016/j.trstmh.2011.01.003

Adriaens, T., Sutton-Croft, M., Owen, K., Brosens, D., Valkenburg, J. van, Kilbey, D., Groom, Q., Ehmig, C., Thürkow, F., Hende, P. V., & Schneider, K. (2015). Trying to engage the crowd in recording invasive alien species in Europe: Experiences from two smartphone applications in northwest Europe. *Management of Biological Invasions, 6*(2), 215–225. https://doi.org/10.3391/mbi.2015.6.2.12

Aigo, J., & Ladio, A. (2016). Traditional Mapuche ecological knowledge in Patagonia, Argentina: Fishes and other living beings inhabiting continental waters, as a reflection of processes of change. *Journal of Ethnobiology and Ethnomedicine, 12*(1), 56. https://doi.org/10.1186/s13002-016-0130-y

Al-Assaf, A., Tadros, M. J., Al-Shishany, S., Stewart, S., Majdalawi, M., Tabieh, M., & Othman, Y. A. (2020). Economic Assessment and Community Management of *Prosopis juliflora* Invasion in Sweimeh Village, Jordan. *Sustainability, 12*(20), Article 20. https://doi.org/10.3390/su12208327

Alavi, A. H., & Buttlar, W. G. (2019). An overview of smartphone technology for citizen-centered, real-time and scalable civil infrastructure monitoring. *Future Generation Computer Systems, 93*, 651–672. https://doi.org/10.1016/j.future.2018.10.059

Allen, D. E. (1976). *The naturalist in Britain: A social history*. Princeton University Press. https://doi.org/10.2307/j.ctv1nxcv20

Alt, C. (2010). *Virginia Woolf and the Study of Nature*. Cambridge University Press.

Amsellem, L., Brouat, C., Duron, O., Porter, S. S., Vilcinskas, A., & Facon, B. (2017). Importance of Microorganisms to Macroorganisms Invasions. In *Networks of Invasion: Empirical Evidence and Case Studies* (Vol. 57, pp. 99–146). Elsevier. https://doi.org/10.1016/bs.aecr.2016.10.005

Archibald, J. L., Anderson, C. B., Dicenta, M., Roulier, C., Slutz, K., & Nielsen, E. A. (2020). The relevance of social imaginaries to understand and manage biological invasions in southern Patagonia. *Biological Invasions, 22*(11), 3307–3323. https://doi.org/10.1007/s10530-020-02325-2

Ascunce, M. S., Yang, C.-C., Oakey, J., Calcaterra, L., Wu, W.-J., Shih, C.-J., Goudet, J., Ross, K. G., & Shoemaker, D. (2011). Global Invasion History of the Fire Ant *Solenopsis invicta*. *Science, 331*(6020), 1066–1068. https://doi.org/10.1126/science.1198734

Atchison, J., & Head, L. (2013). Eradicating Bodies in Invasive Plant Management. *Environment and Planning D: Society and Space, 31*(6), 951–968. https://doi.org/10.1068/d17712

Athayde, S., Stepp, J. R., & Ballester, W. C. (2016). Engaging indigenous and academic knowledge on bees in the Amazon: Implications for environmental management and transdisciplinary research. *Journal of Ethnobiology and Ethnomedicine, 12*(1), 1–19. https://doi.org/10.1186/s13002-016-0093-z

Bach, T. M., Kull, C. A., & Rangan, H. (2019). From killing lists to healthy country: Aboriginal approaches to weed control in the Kimberley, Western Australia. *Journal of Environmental Management, 229*, 182–192. https://doi.org/10.1016/j.jenvman.2018.06.050

Bach, T. M., & Larson, B. M. H. (2017). Speaking About Weeds: Indigenous Elders' Metaphors for Invasive Species and Their Management. *Environmental Values, 26*(5), 561–581. https://doi.org/10.3197/096327117X15002190708119

Bacher, S., Blackburn, T. M., Essl, F., Genovesi, P., Heikkilä, J., Jeschke, J. M., Jones, G., Keller, R., Kenis, M., Kueffer, C., Martinou, A. F., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Richardson, D. M., Roy, H. E., Saul, W.-C., Scalera, R., … Kumschick, S. (2018). Socio-economic impact classification of alien taxa (SEICAT). *Methods in Ecology and Evolution, 9*(1), 159–168. https://doi.org/10.1111/2041-210X.12844

Bailey, S. A., Brown, L., Campbell, M. L., Canning-Clode, J., Carlton, J. T., Castro, N., Chainho, P., Chan, F. T., Creed, J. C., Curd, A., Darling, J., Fofonoff, P., Galil, B. S., Hewitt, C. L., Inglis, G. J., Keith, I., Mandrak, N. E., Marchini, A., McKenzie, C. H., … Zhan, A. (2020). Trends in the detection of aquatic non-indigenous species across global marine, estuarine and freshwater ecosystems: A 50-year perspective. *Diversity and Distributions, 26*(12), 1780–1797. https://doi.org/10.1111/ddi.13167

Balehegn, M. (2016). Ecological and Social Wisdom in Camel Praise Poetry Sung by Afar Nomads of Ethiopia. *Journal of Ethnobiology, 36*(2), 457–472. https://doi.org/10.2993/0278-0771-36.2.457

Banks, N. C., Paini, D. R., Bayliss, K. L., & Hodda, M. (2015). The role of global trade and transport network topology in the human-mediated dispersal of alien species. *Ecology Letters, 18*(2), 188–199. https://doi.org/10.1111/ele.12397

Barbour, W., & Schlesinger, C. (2012). Who's the boss? Post-colonialism, ecological research and conservation management on Australian Indigenous lands. *Ecological Management & Restoration, 13*(1), 36–41. https://doi.org/10.1111/j.1442-8903.2011.00632.x

Beinart, W., & Wotshela, L. (2003). Prickly pear in the Eastern Cape since the 1950s-perspectives from interviews. *Kronos: Journal of Cape History, 28*(1), 191–209. https://www.jstor.org/stable/41056500

Bellard, C., Cassey, P., & Blackburn, T. M. (2016). Alien species as a driver of recent extinctions. *Biology Letters, 12*(2), 20150623. https://doi.org/10.1098/rsbl.2015.0623

Bellard, C., Thuiller, W., Leroy, B., Genovesi, P., Bakkenes, M., & Courchamp, F. (2013). Will climate change promote future invasions? *Global Change Biology, 19*(12), 3740–3748. https://doi.org/10.1111/gcb.12344

Bennett, B. M., & van Sittert, L. (2019). Historicising perceptions and the national management framework for invasive alien plants in South Africa. *Journal of Environmental Management, 229*, 174–181. https://doi.org/10.1016/j.jenvman.2018.07.029

Berhanu, A., & Tesfaye, G. (2006). The *Prosopis* dilemma, impacts on dryland biodiversity and some controlling methods. *Journal of the Drylands, 1*(2), 158–164.

Bertelsmeier, C., & Keller, L. (2018). Bridgehead Effects and Role of Adaptive Evolution in Invasive Populations. *Trends in Ecology & Evolution, 33*(7), 527–534. https://doi.org/10.1016/j.tree.2018.04.014

Bertelsmeier, C., & Ollier, S. (2020). International tracking of the COVID-19 invasion: An amazing example of a globalized scientific coordination effort. *Biological Invasions, 22*(9), 2647–2649. https://doi.org/10.1007/s10530-020-02287-5

Bertolino, S. (2020). Non-native invasive mammal species: Introduction to a themed issue. *Mammal Review, 50*(2), 121–123. https://doi.org/10.1111/mam.12187

Besek, J. (2019). Invasive uncertainties: Environmental change and the politics of limited science. *Environmental Sociology, 5*(4), 416–427. https://doi.org/10.1080/23251042.2019.1624002

Bewell, A. (2017). *Natures in translation: Romanticism and colonial natural history. JHU Press.*

Bhattacharyya, J., & Larson, B. M. H. (2014). The Need for Indigenous Voices in Discourse about Introduced Species: Insights from a Controversy over Wild Horses. *Environmental Values, 23*(6), 663–684. https://doi.org/10.3197/096327114X13947900181031

Bhattacharyya, J., & Slocombe, S. (2017). Animal agency: Wildlife management from a kincentric perspective. *Ecosphere, 8*(10). https://doi.org/10.1002/ecs2.1978

Blackburn, T. M., Cassey, P., & Duncan, R. P. (2020). Colonization pressure: A second null model for invasion biology. *Biological Invasions, 22*(4), 1221–1233. https://doi.org/10.1007/s10530-019-02183-7

Blackburn, T. M., Cassey, P., & Lockwood, J. L. (2009). The role of species traits in the establishment success of exotic birds. *Global Change Biology, 15*(12), 2852–2860. https://doi.org/10.1111/j.1365-2486.2008.01841.x

Blackburn, T. M., Prowse, T. A. A., Lockwood, J. L., & Cassey, P. (2011). Passerine introductions to New Zealand support a positive effect of propagule pressure on establishment success. *Biodiversity and Conservation, 20*(10), 2189–2199. https://doi.org/10.1007/s10531-011-0081-5

Blackburn, T. M., Pyšek, P., Bacher, S., Carlton, J. T., Duncan, R. P., Jarošík, V., Wilson, J. R. U., & Richardson, D. M. (2011). A proposed unified framework for biological invasions. *Trends in Ecology & Evolution, 26*(7), 333–339. https://doi.org/10.1016/j.tree.2011.03.023

Bojko, J., Burgess, A. L., Baker, A. G., & Orr, C. H. (2021). Invasive Non-Native Crustacean Symbionts: Diversity and Impact. *Journal of Invertebrate Pathology, 186*, 107482. https://doi.org/10.1016/j.jip.2020.107482

Bonebrake, T. C., Guo, F., Dingle, C., Baker, D. M., Kitching, R. L., & Ashton, L. A. (2019). Integrating Proximal and Horizon Threats to Biodiversity for Conservation. *Trends in Ecology & Evolution, 34*(9), 781–788. https://doi.org/10.1016/j.tree.2019.04.001

Borrett, S. R., Moody, J., & Edelmann, A. (2014). The rise of network ecology: Maps of the topic diversity and scientific collaboration. *Ecological Modelling, 293*, 111–127. https://doi.org/10.1016/j.ecolmodel.2014.02.019

Bortolus, A., Carltonb, J. T., & Schwindta, E. (2015). Appendix S1: Chronology of the Earliest records of *Spartina alterniflora* in South America and naturalistic details. *Diversity and Distributions, 21*(11), 1267–1283. https://www.researchgate.net/profile/Alejandro-Bortolus/publication/301561625_ddi12377-sup-0001-AppendixS1_RG/links/571a1dd708ae408367bc8216/ddi12377-sup-0001-AppendixS1-RG.pdf

Bowman, D. M. J. S., Balch, J., Artaxo, P., Bond, W. J., Cochrane, M. A., D'Antonio, C. M., DeFries, R., Johnston, F. H., Keeley, J. E., Krawchuk, M. A., Kull, C. A., Mack, M., Moritz, M. A., Pyne, S., Roos, C. I., Scott, A. C., Sodhi, N. S., & Swetnam, T. W. (2011). The human dimension of fire regimes on Earth. *Journal of Biogeography, 38*(12), 2223–2236. https://doi.org/10.1111/j.1365-2699.2011.02595.x

Bowman, D. M. J. S., Kolden, C. A., Abatzoglou, J. T., Johnston, F. H., van der Werf, G. R., & Flannigan, M. (2020). Vegetation fires in the Anthropocene. *Nature Reviews Earth & Environment, 1*(10), 500–515. https://doi.org/10.1038/s43017-020-0085-3

Brady, O. J., & Hay, S. I. (2020). The Global Expansion of Dengue: How *Aedes aegypti* Mosquitoes Enabled the First Pandemic Arbovirus. *Annual Review of Entomology, 65*(1), 191–208. https://doi.org/10.1146/annurev-ento-011019-024918

Braga, R. R., Gómez-Aparicio, L., Heger, T., Vitule, J. R. S., & Jeschke, J. M. (2018). Structuring evidence for invasional meltdown: Broad support but with biases and gaps. *Biological Invasions, 20*(4), 923–936. https://doi.org/10.1007/s10530-017-1582-2

Brauman, K. A., Garibaldi, L. A., Polasky, S., Zayas, C. N., Aumeeruddy Thomas, Y., Brancalion, P., Declerck, F., Mastrangelo, M., Nkongolo, N., Palang, H., Shannon, L., Verma, M., & Shrestha, U. B. (2020). *IPBES Global Assessment Report on Biodiversity and Ecosystem Services Chapter 2.3. Status and Trends -Nature's Contributions to People (NCP)*. Zenodo. https://doi.org/10.5281/ZENODO.3832035

Briski, E., Chan, F. T., Darling, J. A., Lauringson, V., MacIsaac, H. J., Zhan, A., & Bailey, S. A. (2018). Beyond propagule pressure: Importance of selection during the transport stage of biological invasions. *Frontiers in Ecology and the Environment, 16*(6), 345–353. https://doi.org/10.1002/fee.1820

Brondizio, E., Diaz, S., Settele, J., Ngo, H. T., Gueze, M., Aumeeruddy-Thomas, Y., Bai, X., Geschke, A., Molnár, Z., Niamir, A., Pascual, U., Simcock, A., & Jaureguiberry, P. (2019). IPBES Global Assessment Report on Biodiversity and Ecosystem Services Chapter 1 Assessing a planet in transformation: Rationale and approach. In *IPBES Global Assessment on Biodiversity and Ecosystem Services.* Zenodo. https://doi.org/10.5281/ZENODO.3831852

Brooks, M. L., D'Antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., Hobbs, R. J., Pellant, M., & Pyke, D. (2004). Effects of Invasive Alien Plants on Fire Regimes. *BioScience, 54*(7), Article 7. https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2

Brundu, G., & Richardson, D. M. (2016). Planted forests and invasive alien trees in Europe: A code for managing existing and future plantings to mitigate the risk of negative impacts from invasions. *NeoBiota, 30*, 5–47. https://doi.org/10.3897/neobiota.30.7015

Byun, C., de Blois, S., & Brisson, J. (2018). Management of invasive plants through ecological resistance. *Biological Invasions, 20*(1), 13–27. https://doi.org/10.1007/s10530-017-1529-7

CABI. (2022). *Invasive Species Compendium.* Wallingford, UK: CAB International. https://www.cabi.org/ISC

Cacabelos, E., Martins, G. M., Faria, J., Prestes, A. C. L., Costa, T., Moreu, I., &

Neto, A. I. (2020). Limited effects of marine protected areas on the distribution of invasive species, despite positive effects on diversity in shallow-water marine communities. *Biological Invasions, 22*(3), 1169–1179. https://doi.org/10.1007/s10530-019-02171-x

Cadotte, M. W., Dinnage, R., & Tilman, D. (2012). Phylogenetic diversity promotes ecosystem stability. *Ecology, 93*(sp8), S223–S233. https://doi.org/10.1890/11-0426.1

Caluya, G. (2014). Fragments for a postcolonial critique of the Anthropocene: Invasion biology and environmental security. In *Rethinking invasion ecologies from the environmental humanities* (1st ed., pp. 49–62). Routledge.

Cannone, N., Malfasi, F., Favero-Longo, S. E., Convey, P., & Guglielmin, M. (2022). Acceleration of climate warming and plant dynamics in Antarctica. *Current Biology, 32*(7), 1599-1606.e2. https://doi.org/10.1016/j.cub.2022.01.074

Capinha, C., Essl, F., Seebens, H., Moser, D., & Pereira, H. M. (2015). The dispersal of alien species redefines biogeography in the Anthropocene. *Science, 348*(6240), 1248–1251. https://doi.org/10.1126/science.aaa8913

Capinha, C., Rödder, D., Pereira, H. M., & Kappes, H. (2014). Response of non-native European terrestrial gastropods to novel climates correlates with biogeographical and biological traits: Climate-matching and species traits. *Global Ecology and Biogeography, 23*(8), 857–866. https://doi.org/10.1111/geb.12176

Caplat, P., Edelaar, P., Dudaniec, R. Y., Green, A. J., Okamura, B., Cote, J., Ekroos, J., Jonsson, P. R., Löndahl, J., Tesson, S. V., & Petit, E. J. (2016). Looking beyond the mountain: Dispersal barriers in a changing world. *Frontiers in Ecology and the Environment, 14*(5), 261–268. https://doi.org/10.1002/fee.1280

Carlton, J. T. (1992). Introduced marine and estuarine mollusks of North America: An end-of-the-20th-century perspective. *Journal of Shellfish Research, 11*(2), 489–505. https://www.researchgate.net/publication/255642084_Introduced_marine_and_estuarine_mollusks_of_North_America_an_end-of-the-20th-century_perspective

Carter, L., Mankad, A., Zhang, A., Curnock, M. I., & Pollard, C. R. J. (2021). A multidimensional framework to inform stakeholder engagement in the science and management of invasive and pest animal species. *Biological Invasions, 23*(2), 625–640. https://doi.org/10.1007/s10530-020-02391-6

Cassey, P., Delean, S., Lockwood, J. L., Sadowski, J. S., & Blackburn, T. M. (2018). Dissecting the null model for biological invasions: A meta-analysis of the propagule pressure effect. *PLoS Biology, 16*(4), e2005987. https://doi.org/10.1371/journal.pbio.2005987

Catford, J. A., Daehler, C. C., Murphy, H. T., Sheppard, A. W., Hardesty, B. D., Westcott, D. A., Rejmánek, M., Bellingham, P. J., Pergl, J., Horvitz, C. C., & Hulme, P. E. (2012). The intermediate disturbance hypothesis and plant invasions: Implications for species richness and management. *Perspectives in Plant Ecology, Evolution and Systematics, 14*(3), 231–241. https://doi.org/10.1016/j.ppees.2011.12.002

Catford, J. A., Jansson, R., & Nilsson, C. (2009). Reducing redundancy in invasion ecology by integrating hypotheses into a single theoretical framework. *Diversity and Distributions, 15*(1), 22–40. https://doi.org/10.1111/j.1472-4642.2008.00521.x

Cavender-Bares, J., Kozak, K. H., Fine, Paul. V. A., & Kembel, S. W. (2009). The merging of community ecology and phylogenetic biology. *Ecology Letters, 12*(7), 693–715. https://onlinelibrary.wiley.com/doi/10.1111/j.1461-0248.2009.01314.x

CBD. (2002). *Alien species that threaten ecosystems, habitats or species.* UNEP/CBD/COP/6/20. https://www.cbd.int/doc/decisions/cop-06-dec-23-en.pdf

CBD. (2014). *Pathways of introduction of invasive species, their prioritization and management.* 18. https://www.cbd.int/doc/meetings/sbstta/sbstta-18/official/sbstta-18-09-add1-en.pdf

CBD. (2017). *Guidance on integrating biodiversity considerations into One Health approaches* (Subsidiary Body on Scientific, Technical and Technological Advice, Twenty-First Meeting CBD/SBSTTA/21/9; p. 17). UNEP. https://www.cbd.int/doc/c/8e34/8c61/a535d23833e68906c8c7551a/sbstta-21-09-en.pdf

CBD. (2010). *Goals and Sub-targets.* Secretariat of the Convention on Biological Diversity. https://www.cbd.int/2010-target/goals-targets.shtml

CBD. (2020). *Strategic Plan for Biodiversity 2011-2020, including Aichi Biodiversity Targets.* Convention on Biological Diversity; Secretariat of the Convention on Biological Diversity. https://www.cbd.int/sp/

CBD. (2022). 15/4. Kunming-Montreal Global Biodiversity Framework. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity.* Fifteenth meeting (Part II) of Conference of the parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf

Chang, E. H. (2019). *Novel Cultivations: Plants in British Literature of the Global Nineteenth Century* (Vol. 3). University of Virginia Press.

Chapman, D. S., Gunn, I. D. M., Pringle, H. E. K., Siriwardena, G. M., Taylor, P., Thackeray, S. J., Willby, N. J., & Carvalho, L. (2020). Invasion of freshwater ecosystems is promoted by network connectivity to hotspots of human activity. *Global Ecology and Biogeography, 29*(4), 645–655. https://doi.org/10.1111/geb.13051

Chinchio, E., Crotta, M., Romeo, C., Drewe, J. A., Guitian, J., & Ferrari, N. (2020). Invasive alien species and disease risk: An open challenge in public and animal health. *PLoS Pathogens, 16*(10), e1008922. https://doi.org/10.1371/journal.ppat.1008922

Chornesky, E. A., Bartuska, A. M., Aplet, G. H., Britton, K. O., Cummings-Carlson, J., Davis, F. W., Eskow, J., Gordon, D. R., Gottschalk, K. W., Haack, R. A., Hansen, A. J., Mack, R. N., Rahel, F. J., Shannon, M. A., Wainger, L. A., & Wigley, T. B. (2005). Science Priorities for Reducing the Threat of Invasive Species to Sustainable Forestry. *BioScience, 55*(4), 335. https://doi.org/10.1641/0006-3568(2005)055[0335:SPFRTI]2.0.CO;2

Clavero, M. (2014). Shifting Baselines and the Conservation of Non-Native Species: Introduced Species and Baseline Shifts. *Conservation Biology, 28*(5), 1434–1436. https://doi.org/10.1111/cobi.12266

Clergeau, P., & Nuñez, M. A. (2006). The Language of Fighting Invasive Species. *Science, 311*(5763), 951–951. https://doi.org/10.1126/science.311.5763.951b

Clout, M. N., & De Poorter, M. (2005). International Initiatives Against Invasive Alien Species. *Weed Technology, 19*(3), 523–527. https://doi.org/10.1614/WT-04-126.1

Cóbar-Carranza, A. J., García, R. A., Pauchard, A., & Peña, E. (2014). Effect of *Pinus contorta* invasion on forest fuel properties and its potential implications on the fire regime of *Araucaria araucana* and *Nothofagus antarctica* forests. *Biological Invasions, 16*(11), 2273–2291. https://doi.org/10.1007/s10530-014-0663-8

Colautti, R. I., Grigorovich, I. A., & MacIsaac, H. J. (2006). Propagule Pressure: A Null Model for Biological Invasions. Biological Invasions, 8(5), 1023–1037. https://doi.org/10.1007/s10530-005-3735-y

Colautti, R. I., & MacIsaac, H. J. (2004). A neutral terminology to define 'invasive' species. *Diversity and Distributions, 10*(2), Article 2. https://doi.org/10.1111/j.1366-9516.2004.00061.x

Collaboration for Environmental Evidence. (2013). *Guidelines for Systematic Review and Evidence Synthesis in Environmental Management* (Version 4.2). Environmental Evidence: https://environmentalevidence.org/wp-content/uploads/2014/06/Review-guidelines-version-4.2-finalPRINT.pdf

Conn, D. B. (2014). Aquatic invasive species and emerging infectious disease threats: A One Health perspective. *Aquatic Invasions, 9*(3), 383–390. https://doi.org/10.3391/ai.2014.9.3.12

Consortium, S. (2013). Green paper on citizen science. Citizen Science for Europe. *Towards a Better Society of Empowered Citizens and Enhanced Research. Brussels.* https://www.researchgate.net/publication/259230549_Green_Paper_on_Citizen_Science

Cope, R. C., Ross, J. V., Wittmann, T. A., Watts, M. J., & Cassey, P. (2019). Predicting the Risk of Biological Invasions Using Environmental Similarity and Transport Network Connectedness. *Risk Analysis, 39*(1), 35–53. https://doi.org/10.1111/risa.12870

Corrales, X., Coll, M., Ofir, E., Heymans, J. J., Steenbeek, J., Goren, M., Edelist, D., & Gal, G. (2018). Future scenarios of marine resources and ecosystem conditions in the Eastern Mediterranean under the impacts of fishing, alien species and sea warming. *Scientific Reports, 8*(1), 14284. https://doi.org/10.1038/s41598-018-32666-x

Courchamp, F., Chapuis, J.-L., & Pascal, M. (2003). Mammal invaders on islands: Impact, control and control impact. *Biological Reviews, 78*(3), 347–383. https://doi.org/10.1017/S1464793102006061

Coutts, S. R., Helmstedt, K. J., & Bennett, J. R. (2018). Invasion lags: The stories we tell ourselves and our inability to infer process from pattern. *Diversity and Distributions, 24*(2), 244–251. https://doi.org/10.1111/ddi.12669

Cronon, W. (1992). A place for stories: Nature, history, and narrative. *The Journal of American History, 78*(4), 1347–1376. https://www.jstor.org/stable/2079346

Crosby, A. W. (1972). The Columbian Exchange: Biological and Cultural Consequences of 1492. *In Contributions in American studies* (p. 268). Greenwood Publishing Group.

Crosby, A. W. (1986). *Ecological Implerialism: The biological expansion of Europe, 900-1900* (1st ed.). https://www.dynamosys.com/sites/default/files/webform/ecological-imperialism-the-biological-expansion-of-europe-900-1-alfred-w-crosby-2618860.pdf

Crowley, S. L. (2014). Camels Out of Place and Time: The Dromedary *(Camelus dromedarius)* in Australia. *Anthrozoös, 27*(2), 191–203. https://doi.org/10.2752/175303714X13903827487449

Crutzen, P. J., & Stoermer, E. F. (2000). The Anthropocene. *Global Change News Letter, 41*, 17–18. igbp.net/download/18.316f18321323470177580001401/1376383088452/NL41.pdf

Damasceno, G., & Fidelis, A. (2020). Abundance of invasive grasses is dependent on fire regime and climatic conditions in tropical savannas. *Journal of Environmental Management, 271*, 111016. https://doi.org/10.1016/j.jenvman.2020.111016

Dandy, N., Marzano, M., Porth, E. F., Urquhart, J., & Potter, C. (2017). Who has a stake in ash dieback? A conceptual framework for the identification and categorisation of tree health stakeholders. *Swedish University of Agricultural Sciences*, 15–26. https://www.cabi.org/isc/abstract/20183360941

Dasgupta, P. (2021). *The Economics of Biodiversity:The Dasgupta Review. Abridged Version* (p. 103). https://www.gov.uk/government/publications/the-economics-of-biodiversity-the-dasgupta-review-abridged-version-translations

Daszak, P., Cunningham, A. A., & Hyatt, A. D. (2000). Emerging infectious diseases of wildlife—Threats to biodiversity and human health. *Science, 287*(5452), 443–449. https://doi.org/10.1126/science.287.5452.443

Dawkins, M. S. (2017). Animal welfare with and without consciousness. *Journal of Zoology, 301*(1), 1–10. https://doi.org/10.1111/jzo.12434

Dawson, W., Moser, D., van Kleunen, M., Kreft, H., Pergl, J., Pyšek, P., Weigelt, P., Winter, M., Lenzner, B., Blackburn, T. M., Dyer, E. E., Cassey, P., Scrivens, S. L., Economo, E. P., Guénard, B., Capinha, C., Seebens, H., García-Díaz, P., Nentwig, W., … Essl, F. (2017). Global hotspots and correlates of alien species richness across taxonomic groups. *Nature Ecology & Evolution, 1*(7), 0186. https://doi.org/10.1038/s41559-017-0186

de Neergaard, A., Saarnak, C., Hill, T., Khanyile, M., Berzosa, A. M., & Birch-Thomsen, T. (2005). Australian wattle species in the Drakensberg region of South Africa – An invasive alien or a natural resource? *Agricultural Systems, 85*(3), 216–233. https://doi.org/10.1016/j.agsy.2005.06.009

Dehnen-Schmutz, K., Boivin, T., Essl, F., Groom, Q. J., Harrison, L., Touza, J. M., & Bayliss, H. (2018). Alien futures: What is on the horizon for biological invasions? *Diversity and Distributions, 24*(8), 1149–1157. https://doi.org/10.1111/ddi.12755

Diagne, C., Leroy, B., Gozlan, R. E., Vaissière, A.-C., Assailly, C., Nuninger, L., Roiz, D., Jourdain, F., Jarić, I., & Courchamp, F. (2020). InvaCost, a public database of the economic costs of biological invasions worldwide. *Scientific Data, 7*(1), 277. https://doi.org/10.1038/s41597-020-00586-z

Diagne, C., Leroy, B., Vaissière, A.-C., Gozlan, R. E., Roiz, D., Jarić, I., Salles, J.-M., Bradshaw, C. J. A., & Courchamp, F. (2021). High and rising economic costs of biological invasions worldwide. *Nature, 592*(7855), 571–576. https://doi.org/10.1038/s41586-021-03405-6

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J. R., Arico, S., Báldi, A., Bartuska, A., Baste, I. A., Bilgin, A., Brondizio, E., Chan, K. M., Figueroa, V. E., Duraiappah, A., Fischer, M., Hill, R., … Zlatanova, D. (2015). The IPBES Conceptual Framework—Connecting nature and people. *Current Opinion in Environmental Sustainability, 14*, 1–16. https://doi.org/10.1016/j.cosust.2014.11.002

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., Hill, R., Chan, K. M. A., Baste, I. A., Brauman, K. A., Polasky, S., Church, A., Lonsdale, M., Larigauderie, A., Leadley, P. W., van Oudenhoven, A. P. E., van der Plaat, F., Schröter, M., Lavorel, S., … Shirayama, Y. (2018). An inclusive approach to assesse nature's contributions to people. *Science, 359*(6373), 270–272. https://doi.org/10.1126/science.aap8826

Diez, J. M., Sullivan, J. J., Hulme, P. E., Edwards, G., & Duncan, R. P. (2008). Darwin's naturalization conundrum:

Dissecting taxonomic patterns of species invasions. *Ecology Letters, 11*(7), 674–681. https://doi.org/10.1111/j.1461-0248.2008.01178.x

Dizard, Jan. E. (2010). Uneasy Relationships Between Ecology, History, and Restoration. In *Restoration and History: The Search for a Usable Environmental Past* (pp. 172–181). Routledge. https://doi.org/10.4324/9780203860373-24

Dlugosch, K. M., Anderson, S. R., Braasch, J., Cang, F. A., & Gillette, H. D. (2015). The devil is in the details: Genetic variation in introduced populations and its contributions to invasion. *Molecular Ecology, 24*(9), 2095–2111. https://doi.org/10.1111/mec.13183

Doherty, T. S., & Russell, J. C. (2019). Ethical Dimensions of Invasive Animal Management. In *Routledge Handbook of Animal Ethics*. Routledge. https://www.taylorfrancis.com/chapters/edit/10.4324/9781315105840-38/ethical-dimensions-invasive-animal-management-tim-doherty-james-russell

Drake, D. R., Mulder, C. P. H., Towns, D. R., & Daugherty, C. H. (2002). The biology of insularity: An introduction. *Journal of Biogeography, 29*(5–6), 563–569. https://doi.org/10.1046/j.1365-2699.2002.00706.x

Drake, J. A., Mooney, H. A., Di Castri, F., Groves, R. H., Kruger, F. J., Rejmanek, M., & Williamson, M. (Eds.). (1989). *Biological Invasions. A global perspective.* Scientific Committee on Problems of the Environment (SCOPE) of the International Council of Scientific Unions (ICSU). https://pdfs.semanticscholar.org/efd2/a59cccf35f8ec4b14f74b7b830838bbf6fb9.pdf

Duncan, R. P., Blackburn, T. M., Rossinelli, S., & Bacher, S. (2014). Quantifying invasion risk: The relationship between establishment probability and founding population size. *Methods in Ecology and Evolution, 5*(11), 1255–1263. https://doi.org/10.1111/2041-210X.12288

Duncan, R. P., Blackburn, T. M., & Sol, D. (2003). The Ecology of Bird Introductions. *Annual Review of Ecology, Evolution, and Systematics, 34*(1), 71–98. https://doi.org/10.1146/annurev.ecolsys.34.011802.132353

Dunn, A. M., & Hatcher, M. J. (2015). Parasites and biological invasions: Parallels, interactions, and control. *Trends in Parasitology, 31*(5), 189–199. https://doi.org/10.1016/j.pt.2014.12.003

Dyer, E. E., Cassey, P., Redding, D. W., Collen, B., Franks, V., Gaston, K. J., Jones, K. E., Kark, S., Orme, C. D. L., & Blackburn, T. M. (2017). The global distribution and drivers of alien bird species richness. *PLoS Biology, 15*(1), e2000942. https://doi.org/10.1371/journal.pbio.2000942

Early, R., & Sax, D. F. (2014). Climatic niche shifts between species' native and naturalized ranges raise concern for ecological forecasts during invasions and climate change. *Global Ecology and Biogeography, 23*(12), 1356–1365. https://doi.org/10.1111/geb.12208

Ehrenfeld, J. G. (2010). Ecosystem Consequences of Biological Invasions. *Annual Review of Ecology, Evolution, and Systematics, 41*(1), 59–80. https://doi.org/10.1146/annurev-ecolsys-102209-144650

Ellen, R. F. (2020). *The Nuaulu World of Plants: Ethnobotanical Cognition, Knowledge and Practice Among a People of Seram, eastern Indonesia* (Vol. 3). Sean Kingston. https://kar.kent.ac.uk/82526/

Encarnação, J., Teodósio, M. A., & Morais, P. (2021). Citizen Science and Biological Invasions: A Review. *Frontiers in Environmental Science, 8*. https://doi.org/10.3389/fenvs.2020.602980

Ens, E., Daniels, C., Nelson, E., Roy, J., & Dixon, P. (2016). Creating multi-functional landscapes: Using exclusion fences to frame feral ungulate management preferences in remote Aboriginal-owned northern Australia. *Biological Conservation, 197*, 235–246. https://doi.org/10.1016/j.biocon.2016.03.007

Essl, F., Bacher, S., Blackburn, T. M., Booy, O., Brundu, G., Brunel, S., Cardoso, A.-Cristina., Eschen, R., Gallardo, B., Galil, B., García-Berthou, E., Genovesi, P., Groom, Q., Harrower, C., Hulme, P. E., Katsanevakis, S., Kenis, M., Kühn, I., Kumschick, S., … Jeschke, J. M. (2015). Crossing Frontiers in Tackling Pathways of Biological Invasions. *BioScience, 65*(8), 769–782. https://doi.org/10.1093/biosci/biv082

Essl, F., Bacher, S., Genovesi, P., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kowarik, I., Kühn, I., Pyšek, P., Rabitsch, W., Schindler, S., van Kleunen, M., Vilà, M., Wilson, J. R. U., & Richardson, D. M. (2018). Which taxa are alien? Criteria, applications, and uncertainties. *BioScience, 68*(7), 496–509. https://doi.org/10.1093/biosci/biy057

Essl, F., Dullinger, S., Genovesi, P., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kühn, I., Lenzner, B., Pauchard, A., Pyšek, P., Rabitsch, W., Richardson, D. M., Seebens, H., Van Kleunen, M., Van Der Putten, W. H., Vilà, M., & Bacher, S. (2019). A Conceptual Framework for Range-Expanding Species that Track Human-Induced Environmental Change. *BioScience, 69*(11), 908–919. https://doi.org/10.1093/biosci/biz101

Essl, F., Dullinger, S., Rabitsch, W., Hulme, P. E., Hülber, K., Jarošík, V., Kleinbauer, I., Krausmann, F., Kühn, I., Nentwig, W., Vilà, M., Genovesi, P., Gherardi, F., Desprez-Loustau, M.-L., Roques, A., & Pyšek, P. (2011). Socioeconomic legacy yields an invasion debt. *Proceedings of the National Academy of Sciences, 108*(1), 203–207. https://doi.org/10.1073/pnas.1011728108

Essl, F., Lenzner, B., Bacher, S., Bailey, S., Capinha, C., Daehler, C., Dullinger, S., Genovesi, P., Hui, C., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kühn, I., Leung, B., Liebhold, A., Liu, C., MacIsaac, H. J., Meyerson, L. A., Nuñez, M. A., … Roura-Pascual, N. (2020). Drivers of future alien species impacts: An expert-based assessment. *Global Change Biology, 26*(9), 4880–4893. https://doi.org/10.1111/gcb.15199

Estévez, R. A., Anderson, C. B., Pizarro, J. C., & Burgman, M. A. (2015). Clarifying values, risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species management. *Conservation Biology, 29*(1), 19–30. https://doi.org/10.1111/cobi.12359

Estoup, A., & Guillemaud, T. (2010). Reconstructing routes of invasion using genetic data: Why, how and so what? *Molecular Ecology, 19*(19), 4113–4130. https://doi.org/10.1111/j.1365-294X.2010.04773.x

Estoup, A., Ravigné, V., Hufbauer, R., Vitalis, R., Gautier, M., & Facon, B. (2016). Is There a Genetic Paradox of Biological Invasion? *Annual Review of Ecology, Evolution, and Systematics, 47*(1), 51–72. https://doi.org/10.1146/annurev-ecolsys-121415-032116

Estrada-Peña, A., Ostfeld, R. S., Peterson, A. T., Poulin, R., & de la Fuente, J. (2014). Effects of environmental change on zoonotic disease risk: An ecological primer. *Trends in Parasitology, 30*(4), 205–214. https://doi.org/10.1016/j.pt.2014.02.003

European Commission. (2013). *Flash Eurobarometer 379—Attitute towards biodiversity*. https://ec.europa.eu/commfrontoffice/publicopinion/flash/fl_379_en.pdf

European Commission. (2015). *Special Eurobarometer 436—Attitudes of Europeans towards Biodiversity.* European Union. https://ec.europa.eu/COMMFrontOffice/publicopinion/index.cfm/Survey/getSurveyDetail/instruments/SPECIAL/surveyKy/2091

European Commission. (2019). *Attitudes of Europeans towards Biodiversity (2194 / 481)* (Biodiversity). https://europa.eu/eurobarometer/surveys/detail/2194

European Union. (2014). *Regulation (EU) No 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species*. Official Journal of the European Union, 57, 35. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=OJ:JOL_2014_317_R_0003

Evans, E. A. (2003). Economic Dimensions of Invasive Species. *Choices, 18*(2), 5–9. https://www.jstor.org/stable/choices.18.2.0005

Ewel, J. J., & Putz, F. E. (2004). A place for alien species in ecosystem restoration. *Frontiers in Ecology and the Environment, 2*(7), 354–360. https://doi.org/10.1890/1540-9295(2004)002[0354:APFASI]2.0.CO;2

Facon, B., Genton, B. J., Shykoff, J., Jarne, P., Estoup, A., & David, P. (2006). A general eco-evolutionary framework for understanding bioinvasions. *Trends in Ecology & Evolution, 21*(3), 130–135. https://doi.org/10.1016/j.tree.2005.10.012

Fall, J. J. (2013). Biosecurity and ecology: Beyond the nativism debate. In A. Dobson, K. Barker, & S. L. Taylor (Eds.), *Biosecurity: The Socio-Politics of Invasive Species and Infectious Diseases* (pp. 167–182). Earthscan. https://doi.org/10.4324/9780203113110

FAO. (2007). *FAO biosecurity toolkit.* Food & Agriculture Org. https://www.fao.org/3/a1140e/a1140e.pdf

Finderup Nielsen, T., Sand-Jensen, K., Dornelas, M., & Bruun, H. H. (2019). More is less: Net gain in species richness, but biotic homogenization over 140 years. *Ecology Letters, 22*(10), 1650–1657. https://doi.org/10.1111/ele.13361

Fischer, J. R. (2007). Cattle in Hawai'i: Biological and Cultural Exchange. *Pacific Historical Review, 76*(3), 347–372. https://doi.org/10.1525/phr.2007.76.3.347

Fish, J., Chiche, Y., Day, R., Efa, N., Witt, A., Fessehaie, R., De Graft Johnson, K., Gumisizira, G., & Nkandu, B. (2010). *Mainstreaming gender into prevention and management of invasive species.* Global Invasive Species Programme (GISP). https://portals.iucn.org/library/node/9837

Fitzpatrick, M. C., Weltzin, J. F., Sanders, N. J., & Dunn, R. R. (2007). The biogeography of prediction error: Why does the introduced range of the fire ant over-predict its native range? *Global Ecology and Biogeography, 16*(1), 24–33. https://doi.org/10.1111/j.1466-8238.2006.00258.x

Follett, R., & Strezov, V. (2015). An Analysis of Citizen Science Based Research: Usage and Publication Patterns. *PLoS ONE, 10*(11), e0143687. https://doi.org/10.1371/journal.pone.0143687

Foster, B. L., Houseman, G. R., Hall, D. R., & Hinman, S. E. (2015). Does tallgrass prairie restoration enhance the invasion resistance of post-agricultural lands? *Biological Invasions, 17*(12), 3579–3590. https://doi.org/10.1007/s10530-015-0979-z

Foxcroft, L. C., Pickett, S. T. A., & Cadenasso, M. L. (2011). Expanding the conceptual frameworks of plant invasion ecology. *Perspectives in Plant Ecology, Evolution and Systematics, 13*(2), 89–100. https://doi.org/10.1016/j.ppees.2011.03.004

Franzese, J., & Raffaele, E. (2017). Fire as a driver of pine invasions in the Southern Hemisphere: A review. *Biological Invasions, 19*(8), Article 8. https://doi.org/10.1007/s10530-017-1435-z

Frawley, J., & McCalman, I. (Eds.). (2014). *Rethinking Invasion Ecologies from the Environmental Humanities* (0 ed.). Routledge. https://doi.org/10.4324/9781315879642

Futhazar, G. (2020). The conceptual challenges of invasive alien species to non-human rights. *Journal of Human Rights and the Environment, 11*(2), 224–243. https://doi.org/10.4337/jhre.2020.02.04

Gaertner, M., Biggs, R., Te Beest, M., Hui, C., Molofsky, J., & Richardson, D. M. (2014). Invasive plants as drivers of regime shifts: Identifying high-priority invaders that alter feedback relationships. *Diversity and Distributions, 20*(7), 733–744. https://doi.org/10.1111/ddi.12182

Galil, B. S. (2017). Eyes Wide Shut: Managing Bio-Invasions in Mediterranean Marine Protected Areas. In P. D. Goriup (Ed.), *Management of Marine Protected Areas* (pp. 187–206). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781119075806.ch10

Gallardo, B., Bacher, S., Bradley, B., Comín, F. A., Gallien, L., Jeschke, J. M., Sorte, C. J. B., & Vilà, M. (2019). InvasiBES: Understanding and managing the impacts of invasive alien species on biodiversity and ecosystem Services. *NeoBiota, 50*, 109–122. https://doi.org/10.3897/neobiota.50.35466

García-Llorente, M., Martín-López, B., González, J. A., Alcorlo, P., & Montes, C. (2008). Social perceptions of the impacts and benefits of invasive alien species: Implications for management. *Biological Conservation, 141*(12), 2969–2983. https://doi.org/10.1016/j.biocon.2008.09.003

Gardiner, M. M., & Roy, H. E. (2022). The Role of Community Science in Entomology. *Annual Review of Entomology, 67*(1), 437–456. https://doi.org/10.1146/annurev-ento-072121-075258

Garnett, S. T., Burgess, N. D., Fa, J. E., Fernández-Llamazares, Á., Molnár, Z., Robinson, C. J., Watson, J. E. M., Zander, K. K., Austin, B., Brondizio, E. S., Collier, N. F., Duncan, T., Ellis, E., Geyle, H., Jackson, M. V., Jonas, H., Malmer, P., McGowan, B., Sivongxay, A., & Leiper, I. (2018). A spatial overview of the global importance of Indigenous lands for conservation. *Nature Sustainability, 1*(7), 369–374. https://doi.org/10.1038/s41893-018-0100-6

Gautier, M., Vitalis, R., Flori, L., & Estoup, A. (2022). *F* -Statistics estimation and admixture graph construction with Pool-Seq or allele count data using the R package poolfstat. *Molecular Ecology Resources, 22*(4), 1394–1416. https://doi.org/10.1111/1755-0998.13557

Gawith, D., Greenaway, A., Samarasinghe, O., Bayne, K., Velarde, S., & Kravchenko, A. (2020). Socio-ecological mapping generates public understanding of wilding conifer incursion. *Biological Invasions, 22*(10), 3031–3049. https://doi.org/10.1007/s10530-020-02309-2

Gbedomon, R. C., Salako, V. K., & Schlaepfer, M. A. (2020). Diverse views among scientists on non-native species. *NeoBiota, 54*, 49–69. https://doi.org/10.3897/neobiota.54.38741

GBIF. (2021). *GBIF Backbone Taxonomy.* https://doi.org/10.15468/39omei

Genovart, M., Tavecchia, G., Enseñat, J. J., & Laiolo, P. (2013). Holding up a mirror to the society: Children recognize exotic species much more than local ones. *Biological Conservation, 159*, 484–489. https://doi.org/10.1016/j.biocon.2012.10.028

Genovesi, P., Bacher, S., Kobelt, M., Pascal, M., & Scalera, R. (2009). Alien Mammals of Europe. In *Handbook of Alien Species in Europe* (Vol. 3, pp. 119–128). Springer Netherlands. https://doi.org/10.1007/978-1-4020-8280-1_9

Genovesi, P., & Monaco, A. (2013). Guidelines for Addressing Invasive Species in Protected Areas. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (pp. 487–506). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_22

Giakoumi, S., Guilhaumon, F., Kark, S., Terlizzi, A., Claudet, J., Felline, S., Cerrano, C., Coll, M., Danovaro, R., Fraschetti, S., Koutsoubas, D., Ledoux, J., Mazor, T., Mérigot, B., Micheli, F., & Katsanevakis, S. (2016). Space invaders; biological invasions in marine conservation planning. *Diversity and Distributions, 22*(12), 1220–1231. https://doi.org/10.1111/ddi.12491

Goëau, H., Bonnet, P., Joly, A., Bakić, V., Barbe, J., Yahiaoui, I., Selmi, S., Carré, J., Barthélémy, D., Boujemaa, N., Molino, J.-F., Duché, G., & Péronnet, A. (2013). Pl@ntNet mobile app. *Proceedings of the 21st ACM International Conference on Multimedia*, 423–424. https://doi.org/10.1145/2502081.2502251

González-Varo, J. P., Biesmeijer, J. C., Bommarco, R., Potts, S. G., Schweiger, O., Smith, H. G., Steffan-Dewenter, I., Szentgyörgyi, H., Woyciechowski, M., & Vilà, M. (2013). Combined effects of global change pressures on animal-mediated pollination. *Trends in Ecology & Evolution, 28*(9), 524–530. https://doi.org/10.1016/j.tree.2013.05.008

Gratani, M., Butler, J. R. A., Royee, F., Valentine, P., Burrows, D., Canendo, W. I., & Anderson, A. S. (2011). Is Validation of Indigenous Ecological Knowledge a Disrespectful Process? A Case Study of Traditional Fishing Poisons and Invasive Fish Management from the Wet Tropics, Australia. *Ecology and Society, 16*(3), art25. https://doi.org/10.5751/ES-04249-160325

Grenz, J. B. (2020). *Healing the land by reclaiming an Indigenous ecology: A journey exploring the application of the Indigenous worldview to invasion biology and ecology.* https://doi.org/10.14288/1.0394715

Grice, A. C., Cassady, J., & Nicholas, D. M. (2012). Indigenous and non-Indigenous knowledge and values combine to support management of Nywaigi lands in the Queensland coastal tropics. Ecological *Management & Restoration, 13*(1), 93–97. https://doi.org/10.1111/j.1442-8903.2011.00621.x

Griffiths, T., & Robin, L. (Eds.). (1997). *Ecology and empire: Environmental history of settler societies*. Edinburgh University Press. https://www.jstor.org/stable/10.3366/j.ctvxcrw7v

Grove, R. H. (1996). *Green Imperialism: Colonial Expansion, Tropical Island Edens and the Origins of Environmentalism, 1600-1860.* Cambridge University Press. https://books.google.co.jp/books?id=h6xSzhlmNdoC

Gura, T. (2013). Citizen science: Amateur experts. *Nature, 496*(7444), Article 7444. https://doi.org/10.1038/nj7444-259a

Hall, S. A., Bastos, R., Vicente, J., Vaz, A. S., Honrado, J. P., Holmes, P. M., Gaertner, M., Esler, K. J., & Cabral, J. A. (2021). A dynamic modeling tool to anticipate the effectiveness of invasive plant control and restoration recovery trajectories in South African fynbos. *Restoration Ecology, 29*(3), e13324. https://doi.org/10.1111/rec.13324

Hall, S. J. (2009). Cultural Disturbances and Local Ecological Knowledge Mediate Cattail *(Typha domingensis)* Invasion in Lake Pátzcuaro, México. *Human Ecology, 37*(2), 241–249. https://doi.org/10.1007/s10745-009-9228-3

Hallett, L. M., Standish, R. J., Hulvey, K. B., Gardener, M. R., Suding, K. N., Starzomski, B. M., Murphy, S. D., & Harris, J. A. (2013). Towards a Conceptual Framework for Novel Ecosystems. In R. J. Hobbs, E. S. Higgs, & C. M. Hall (Eds.), *Novel Ecosystems* (pp. 16–28). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118354186.ch3

Halpern, B. S., Silliman, B. R., Olden, J. D., Bruno, J. P., & Bertness, M. D. (2007). Incorporating positive interactions in aquatic restoration and conservation. *Frontiers in Ecology and the Environment, 5*(3), 153–160. https://doi.org/10.1890/1540-9295(2007)5[153:IPIIAR]2.0.CO;2

Hampton, J. O., & Hyndman, T. H. (2019). Underaddressed animal-welfare issues in conservation. *Conservation Biology, 33*(4), 803–811. https://doi.org/10.1111/cobi.13267

Haram, L. E., Carlton, J. T., Centurioni, L., Crowley, M., Hafner, J., Maximenko, N., Murray, C. C., Shcherbina, A. Y., Hormann, V., Wright, C., & Ruiz, G. M. (2021). Emergence of a neopelagic community through the establishment of coastal species on the high seas. *Nature Communications, 12*(1), 6885. https://doi.org/10.1038/s41467-021-27188-6

Harfoot, M., Tittensor, D. P., Newbold, T., McInerny, G., Smith, M. J., & Scharlemann, J. P. W. (2014). Integrated assessment models for ecologists: The present and the future. *Global Ecology and Biogeography, 23*(2), 124–143. https://doi.org/10.1111/geb.12100

Harrop, S. R. (2013). Wild Animal Welfare in International Law: The Present Position and the Scope for Development. *Global Policy, 4*(4), 381–390. https://doi.org/10.1111/1758-5899.12086

Harvey, E., Gounand, I., Ward, C. L., & Altermatt, F. (2017). Bridging ecology and conservation: From ecological networks to ecosystem function. *Journal of Applied Ecology, 54*(2), 371–379. https://doi.org/10.1111/1365-2664.12769

Havlík, P., Valin, H., Herrero, M., Obersteiner, M., Schmid, E., Rufino, M. C., Mosnier, A., Thornton, P. K., Böttcher, H., Conant, R. T., Frank, S., Fritz, S., Fuss, S., Kraxner, F., & Notenbaert, A. (2014). Climate change mitigation through livestock system transitions. *Proceedings of the National Academy of Sciences, 111*(10), 3709–3714. https://doi.org/10.1073/pnas.1308044111

Head, L. (2017). The social dimensions of invasive plants. *Nature Plants, 3*(6), 17075. https://doi.org/10.1038/nplants.2017.75

Head, L., & Atchison, J. (2015). Governing invasive plants: Policy and practice in managing the Gamba grass *(Andropogon gayanus)* – Bushfire nexus in northern Australia. *Land Use Policy, 47*, 225–234. https://doi.org/10.1016/j.landusepol.2015.04.009

Head, L., Larson, BrendonM. H., Hobbs, R., Atchison, J., Gill, N., Kull, C., & Rangan, H. (2015). Living with Invasive Plants in the Anthropocene: The Importance of Understanding Practice and Experience. *Conservation and Society, 13*(3), 311. https://doi.org/10.4103/0972-4923.170411

Head, L., & Muir, P. (2004). Nativeness, Invasiveness, and Nation in Australian Plants. *Geographical Review, 94*(2), 199–217. https://doi.org/10.1111/j.1931-0846.2004.tb00167.x

Head, L., Trigger, D., & Mulcock, J. (2005). Culture as Concept and Influence in Environmental Research and Management. *Conservation and Society, 3*(2), 251–264. https://www.jstor.org/stable/26396578

Higgs, E. (2003). *Nature by Design: People, Natural Process, and Ecological Restoration.* MIT Press. https://direct.mit.edu/books/book/2569/Nature-by-DesignPeople-Natural-Process-and

Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., Epstein, P. R., Ewel, J. J., Klink, C. A., Lugo, A. E., Norton, D., Ojima, D., Richardson, D. M., Sanderson, E. W., Valladares, F., Vilà, M., Zamora, R., & Zobel, M. (2006). Novel ecosystems: Theoretical and management aspects of the new ecological world order: Novel ecosystems. *Global Ecology and Biogeography, 15*(1), 1–7. https://doi.org/10.1111/j.1466-822X.2006.00212.x

Hobbs, R. J., Higgs, E. S., & Harris, J. A. (2014). Novel ecosystems: Concept or inconvenient reality? A response to Murcia *et al. Trends in Ecology & Evolution, 29*(12), 645–646. https://doi.org/10.1016/j.tree.2014.09.006

Hovanec, C. (2018). *Animal Subjects: Literature, Zoology, and British Modernism* (Vol. 1). Cambridge University Press. https://www.cambridge.org/us/academic/subjects/literature/english-literature-1900-1945/animal-subjects-literature-zoology-and-british-modernism-volume-1

Howard, P. L. (2019). Human adaptation to invasive species: A conceptual framework based on a case study metasynthesis. *Ambio, 48*(12), 1401–1430. https://doi.org/10.1007/s13280-019-01297-5

Hufbauer, R. A., Facon, B., Ravigné, V., Turgeon, J., Foucaud, J., Lee, C. E., Rey, O., & Estoup, A. (2012). Anthropogenically induced adaptation to invade (AIAI): Contemporary adaptation to human-altered habitats within the native range can promote invasions: Anthropogenically induced adaptation to invade. *Evolutionary Applications, 5*(1), 89–101. https://doi.org/10.1111/j.1752-4571.2011.00211.x

Hui, C., & Richardson, D. M. (2019). How to Invade an Ecological Network. *Trends in Ecology & Evolution, 34*(2), 121–131. https://doi.org/10.1016/j.tree.2018.11.003

Hui, C., Richardson, D. M., Landi, P., Minoarivelo, H. O., Garnas, J., & Roy, H. E. (2016). Defining invasiveness and invasibility in ecological networks. *Biological Invasions, 18*(4), 971–983. https://doi.org/10.1007/s10530-016-1076-7

Hui, C., Richardson, D. M., Landi, P., Minoarivelo, H. O., Roy, H. E., Latombe, G., Jing, X., CaraDonna, P. J., Gravel, D., Beckage, B., & Molofsky, J. (2021). Trait positions for elevated invasiveness in adaptive ecological networks. *Biological Invasions, 23*(6), 1965–1985. https://doi.org/10.1007/s10530-021-02484-w

Hulme, P. E. (2014). Invasive species challenge the global response to emerging diseases. *Trends in Parasitology, 30*(6), 267–270. https://doi.org/10.1016/j.pt.2014.03.005

Hulme, P. E. (2020a). One Biosecurity: A unified concept to integrate human, animal, plant, and environmental health. *Emerging Topics in Life Sciences, 4*(5), 539–549. https://doi.org/10.1042/ETLS20200067

Hulme, P. E. (2020b). Plant invasions in New Zealand: Global lessons in prevention, eradication and control. *Biological Invasions, 22*(5), 1539–1562. https://doi.org/10.1007/s10530-020-02224-6

Hulme, P. E. (2021). Advancing One Biosecurity to Address the Pandemic Risks of Biological Invasions. *BioScience, 71*(7), 708–721. https://doi.org/10.1093/biosci/biab019

Hulme, P. E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., Nentwig, W., Olenin, S., Panov, V., Pergl, J., Pyšek, P., Roques, A., Sol, D., Solarz, W., & Vilà, M. (2008). Grasping at the routes of biological invasions: A framework for integrating pathways into policy. *Journal of Applied Ecology, 45*(2), 403–414. https://doi.org/10.1111/j.1365-2664.2007.01442.x

Hurteau, M. D., Bradford, J. B., Fulé, P. Z., Taylor, A. H., & Martin, K. L. (2014). Climate change, fire management, and ecological services in the southwestern US. *Forest Ecology and Management, 327*, 280–289. https://doi.org/10.1016/j.foreco.2013.08.007

Hurtt, G. C., Chini, L. P., Frolking, S., Betts, R. A., Feddema, J., Fischer, G., Fisk, J. P., Hibbard, K., Houghton, R. A., Janetos, A., Jones, C. D., Kindermann, G., Kinoshita, T., Klein Goldewijk, K., Riahi, K., Shevliakova, E., Smith, S., Stehfest, E., Thomson, A., … Wang, Y. P. (2011). Harmonization of land-use scenarios for the period 1500–2100: 600 years of global gridded annual land-use transitions, wood harvest, and resulting secondary lands. *Climatic Change, 109*(1–2), 117–161. https://doi.org/10.1007/s10584-011-0153-2

Inglis, M. I. (2020). Wildlife Ethics and Practice: Why We Need to Change the Way We Talk About 'Invasive Species.' *Journal of Agricultural and Environmental Ethics, 33*(2), 299–313. https://doi.org/10.1007/s10806-020-09825-0

IPBES. (2016a). *Summary for policymakers of the assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production*. Zenodo. https://doi.org/10.5281/ZENODO.2616458

IPBES. (2016b). *The assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3402857

IPBES. (2016c). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3235428

IPBES. (2018a). *Information on scoping for a thematic assessment of invasive alien species and their control (deliverable 3 (b) (ii)).* https://ipbes.net/sites/default/files/ipbes-6-inf-10_en.pdf

IPBES. (2018b). *IPBES Guide on the Production of Assessments*. (p. 56). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://ipbes.net/guide-production-assessments

IPBES. (2018c). *The IPBES assessment report on land degradation and restoration.* (p. 744). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237392

IPBES. (2018d). *The IPBES regional assessment report on biodiversity and ecosystem services for Africa.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://zenodo.org/record/3236178#.YmjyldrP2Uk

IPBES. (2018e). *The IPBES regional assessment report on biodiversity and ecosystem services for Asia and the Pacific* (Report). Secretariat of the Intergovernmental Science-Policy

Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237374

IPBES. (2018f). *The IPBES regional assessment report on biodiversity and ecosystem services for Europe and Central Asia* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237429

IPBES. (2018g). *The IPBES regional assessment report on biodiversity and ecosystem services for the Americas.* https://ipbes.net/document-library-catalogue/regional-assessment-report-biodiversity-and-ecosystem-services-americas

IPBES. (2019). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3831673

IPBES. (2020a). *IPBES Data Management Policy* (1.1). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://zenodo.org/record/4299999

IPBES. (2020b). *Report of the Indigenous and local knowledge dialogue workshop on the first order draft of the IPBES assessment of invasive alien species* (J. L. Andreve, R. Batzin Chojoj, A. Black, J. T. Cleofe, F. Daguitan, C. Grant, J. A. Guillao, L. Jacobs, T. Malcolm, L. Mulenkei, K. Kumar Rai, A. Nzovu, J. M. Ole Kaunga, M. E. Regpala, N. Sall, P. Shulbaeva, R. Spencer, P. Timoti, & Y. Upun, Eds.). https://ipbes.net/sites/default/files/inline-files/IPBES_IAS_2ndILKDialogue_FOD_Report_FINAL_ForWeb.pdf

IPBES. (2020c). *Workshop Report on Biodiversity and Pandemics of the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES)* (1.3). Zenodo. https://doi.org/10.5281/ZENODO.4147317

IPBES. (2022a). *Methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Zenodo. https://doi.org/10.5281/ZENODO.6522523

IPBES. (2022b). *Report of the third Indigenous and local knowledge dialogue workshop for the IPBES thematic assessment of invasive alien species and their control.* https://ipbes.net/sites/default/files/2023-02/IPBES_IAS_3rdILKDialogue_SPM-SOD_Report_FinalForWeb2.pdf

IPBES. (2022c). *Thematic assessment of the sustainable use of wild species of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Zenodo. https://doi.org/10.5281/ZENODO.6448568

IPBES. (2022d). Foundations of the nature futures framework. *Annex VI to Decision IPBES-9/1.* IPBES 9 Plenary, Bonn, Germany. https://www.ipbes.net/resource-file/104136

IPBES Technical Support Unit On Knowledge And Data. (2021). *IPBES regions and sub-regions* (1.2) [dataset]. Zenodo. https://doi.org/10.5281/ZENODO.5719431

IPCC. (2014). *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects.* (C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White, Eds.; p. 1132). Cambridge University Press. https://doi.org/10.1017/CBO9781107415379

ISAC. (2017). *Advanced Biotechnology Tools for Invasive Species Management.* Invasive Species Advisory Committee. https://www.glc.org/wp-content/uploads/isac_advanced_biotechnology_white_paper.pdf

IUCN. (2000). *UCN Guidelines for the Prevention of Biodiversity Loss Caused by Alien Invasive Species.* (p. 25). https://portals.iucn.org/library/efiles/documents/Rep-2000-052.pdf

IUCN. (2017). *Guidance for interpretation of CBD categories on introduction pathways* [Technical note prepared by IUCN for the European Commission]. International Union for Conservation of Nature. https://nora.nerc.ac.uk/id/eprint/519129/1/N519129CR.pdf

Jackson, M. C. (2015). Interactions among multiple invasive animals. *Ecology, 96*(8), 2035–2041. https://doi.org/10.1890/15-0171.1

Jagoret, P., Michel-Dounias, I., Snoeck, D., Ngnogué, H. T., & Malézieux, E. (2012). Afforestation of savannah with cocoa agroforestry systems: A small-farmer innovation in central Cameroon. *Agroforestry Systems, 86*(3), 493–504. https://doi.org/10.1007/s10457-012-9513-9

Janovsky, R. M., & Larson, E. R. (2019). Does invasive species research use more militaristic language than other ecology and conservation biology literature? *NeoBiota, 44*, 27–38. https://doi.org/10.3897/neobiota.44.32925

Jarić, I., Courchamp, F., Correia, R. A., Crowley, S. L., Essl, F., Fischer, A., González-Moreno, P., Kalinkat, G., Lambin, X., Lenzner, B., Meinard, Y., Mill, A., Musseau, C., Novoa, A., Pergl, J., Pyšek, P., Pyšková, K., Robertson, P., Schmalensee, M., … Jeschke, J. M. (2020). The role of species charisma in biological invasions. *Frontiers in Ecology and the Environment, 18*(6), 345–353. https://doi.org/10.1002/fee.2195

Jeschke, J. M., Keesing, F., & Ostfeld, R. S. (2013). Novel Organisms: Comparing Invasive Species, GMOs, and Emerging Pathogens. *AMBIO, 42*(5), 541–548. https://doi.org/10.1007/s13280-013-0387-5

Jevon, T., & Shackleton, C. M. (2015). Integrating Local Knowledge and Forest Surveys to Assess *Lantana camara* Impacts on Indigenous Species Recruitment in Mazeppa Bay, South Africa. *Human Ecology, 43*(2), 247–254. https://doi.org/10.1007/s10745-015-9748-y

Jones, K. E., Patel, N. G., Levy, M. A., Storeygard, A., Balk, D., Gittleman, J. L., & Daszak, P. (2008). Global trends in emerging infectious diseases. *Nature, 451*(7181), 990–993. https://doi.org/10.1038/nature06536

Kalnicky, E. (2012). *A Coupled Human and Natural Systems Approach to Understanding an Invasive Frog,* Eleutherodactylus coqui, *in Hawaii* [Doctoral thesis, Utah State University]. https://digitalcommons.usu.edu/etd/1412

Kanie, N., & Biermann, F. (Eds.). (2017). *Governing through Goals: Sustainable Development Goals as Governance Innovation.* MIT Press. https://www.lehmanns.de/shop/naturwissenschaften/36547155-9780262035620-governing-through-goals

Kannan, R., Shackleton, C. M., Krishnan, S., & Shaanker, R. U. (2016). Can local use assist in controlling invasive alien species in tropical forests? The case of *Lantana camara* in southern India. *Forest Ecology and Management, 376*, 166–173. https://doi.org/10.1016/j.foreco.2016.06.016

Kapitza, K., Zimmermann, H., Martín-López, B., & Von Wehrden, H. (2019). Research on the social perception of invasive species: A systematic literature review. *NeoBiota, 43*, 47–68. https://doi.org/10.3897/neobiota.43.31619

Keller, S. R., Gilbert, K. J., Fields, P. D., & Taylor, D. R. (2012). Bayesian inference of a complex invasion history revealed by nuclear and chloroplast genetic diversity in the colonizing plant, *Silene latifolia*. *Molecular Ecology, 21*(19), 4721–4734. https://doi.org/10.1111/j.1365-294X.2012.05751.x

Kelly, L. T., Giljohann, K. M., Duane, A., Aquilué, N., Archibald, S., Batllori, E., Bennett, A. F., Buckland, S. T., Canelles, Q., Clarke, M. F., Fortin, M.-J., Hermoso, V., Herrando, S., Keane, R. E., Lake, F. K., McCarthy, M. A., Morán-Ordóñez, A., Parr, C. L., Pausas, J. G., … Brotons, L. (2020). Fire and biodiversity in the Anthropocene. *Science, 370*(6519), eabb0355. https://doi.org/10.1126/science.abb0355

Kelsch, A., Takahashi, Y., Dasgupta, R., Mader, A. D., Johnson, B. A., & Kumar, P. (2020). Invasive alien species and local communities in socio-ecological production landscapes and seascapes: A systematic review and analysis. *Environmental Science & Policy, 112*, 275–281. https://doi.org/10.1016/j.envsci.2020.06.014

Kideys, A. E., & Gücü, A. C. (1995). *Rhopilema Nomadica*: A Lessepsian Scyphomedusan New to the Mediterranean Coast of Turkey. *Israel Journal of Zoology, 41*(4), 615–617. https://doi.org/10.1080/00212210.1995.10688827

Koichi, K., Sangha, K. K., Cottrell, A., & Gordon, I. J. (2012). Aboriginal Rangers' Perspectives on Feral Pigs: Are they a Pest or a Resource? A Case Study in the Wet Tropics World Heritage Area of Northern Queensland. *Journal of Australian Indigenous Issues, 15*(1), 2–20. https://www.researchgate.net/profile/Kamaljit-Sangha-2/publication/260752873_Aboriginal_Rangers'_Perspectives_on_Feral_Pigs_Are_they_a_Pest_or_a_Resource_A_Case_Study_in_the_Wet_Tropics_World_Heritage_Area_of_Northern_Queensland/links/00b7d53224cfdb852f000000/Aboriginal-Rangers-Perspectives-on-Feral-Pigs-Are-they-a-Pest-or-a-Resource-A-Case-Study-in-the-Wet-Tropics-World-Heritage-Area-of-Northern-Queensland.pdf

Kowarik, I. (1995). Time lags in biological invasions with regard to the success and failure of alien species. In P. Pyšek, K. Prach, M. Rejmánek, & M. Wade (Eds.), *Plant invasions: General aspects and special problems* (pp. 15–38). SPB Academic Publishing. https://www.cabi.org/isc/abstract/19952312467

Kueffer, C. (2017). Plant invasions in the Anthropocene. *Science, 358*(6364), 724–725. https://doi.org/10.1126/science.aao6371

Kueffer, C., & Kull, C. A. (2017). Non-native Species and the Aesthetics of Nature. In M. Vilà & P. E. Hulme (Eds.), *Impact of Biological Invasions on Ecosystem Services* (pp. 311–324). Springer International Publishing. https://doi.org/10.1007/978-3-319-45121-3_20

Kueffer, C., Pyšek, P., & Richardson, D. M. (2013). Integrative invasion science: Model systems, multi-site studies, focused meta-analysis and invasion syndromes. *New Phytologist, 200*(3), 615–633. https://doi.org/10.1111/nph.12415

Kull, C. A. (2018). Critical Invasion Science: Weeds, Pests, and Aliens. In R. Lave, C. Biermann, & S. N. Lane (Eds.), *The Palgrave Handbook of Critical Physical Geography* (pp. 249–272). Springer International Publishing. https://doi.org/10.1007/978-3-319-71461-5_12

Kull, C. A., Kueffer, C., Richardson, D. M., Vaz, A. S., Vicente, J. R., & Honrado, J. P. (2018). Using the "regime shift" concept in addressing social–ecological change. *Geographical Research, 56*(1), 26–41. https://doi.org/10.1111/1745-5871.12267

Laforest, J., & Bargeron, C. T. (2011). EDDMapS: The common operating platform for aggregating and using invasive species distribution data. *Phytopathology, 101*(6), S96–S97.

Lambdon, P. W., Pyšek, P., Başnou, C., Hejda, M., Arianoutsou, M., Essl, F., Jarošík, V., Pergl, J., Winter, M., Anastasiu, P., Andriopoulos, P., Bazos, I., Brundu, G., Celesti-Grapow, L., Chassot, P., Delipetrou, P., Josefsson, M., Kark, S., Klotz, S., … Hulme, P. E. (2008). Alien flora of Europe: Species diversity, temporal trends, geographical patterns and research needs. *Preslia, 80*(2), 101–149. http://www.preslia.cz/P082Lam.pdf

Lane, N. D., Miluzzo, E., Lu, H., Peebles, D., Choudhury, T., & Campbell, A. T. (2010). A survey of mobile phone sensing. *IEEE Communications Magazine, 48*(9), 140–150. http://gtubicomp2015grad.pbworks.com/w/file/fetch/94746338/lane-survey-of-mobile-phone-sensing-2010.pdf

Larson, B. M. (2005). The war of the roses: Demilitarizing invasion biology. *Frontiers in Ecology and the Environment, 3*(9), 495–500. https://doi.org/10.1890/1540-9295(2005)003[0495:TWOTRD]2.0.CO;2

Lazzaro, L., Essl, F., Lugliè, A., Padedda, B. M., Pyšek, P., & Brundu, G. (2018). Invasive alien plant impacts on human health and well-being. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health* (pp. 16–33). CABI. https://doi.org/10.1079/9781786390981.0016

Le Maitre, D. C., Gaertner, M., Marchante, E., Ens, E.-J., Holmes, P. M., Pauchard, A., O'Farrell, P. J., Rogers, A. M., Blanchard, R., Blignaut, J., & Richardson, D. M. (2011). Impacts of invasive Australian acacias: Implications for management and restoration: Australian acacias: linking impacts and restoration. *Diversity and Distributions, 17*(5), 1015–1029. https://doi.org/10.1111/j.1472-4642.2011.00816.x

Lenzner, B., Leclère, D., Franklin, O., Seebens, H., Roura-Pascual, N., Obersteiner, M., Dullinger, S., & Essl, F. (2019). A Framework for Global Twenty-First Century Scenarios and Models of Biological Invasions. *BioScience, 69*(9), 697–710. https://doi.org/10.1093/biosci/biz070

Lerner, H., & Berg, C. (2017). A Comparison of Three Holistic Approaches to Health: One Health, EcoHealth, and Planetary Health. *Frontiers in Veterinary Science, 4*, 163. https://doi.org/10.3389/fvets.2017.00163

Leung, B., Roura-Pascual, N., Bacher, S., Heikkila, J., Brotons, L., Burgman, M. A., Dehnen-Schmutz, K., Essl, F., Hulme, P. E., Richardson, D. M., Sol, D., & Vilà, M. (2012). TEASIng apart alien species risk assessments: A framework for best practices. *Ecology Letters, 15*(12), 1475–1493. https://doi.org/10.1111/ele.12003

Levin, S. A. (1992). The problem of pattern and scale in ecology: The Robert H. MacArthur award lecture. *Ecology, 73*(6), Article 6. https://esajournals.onlinelibrary.wiley.com/doi/abs/10.2307/1941447

Li, D., Olden, J. D., Lockwood, J. L., Record, S., McKinney, M. L., & Baiser, B. (2020). Changes in taxonomic and phylogenetic diversity in the Anthropocene. *Proceedings of the Royal Society B, 287*(1929). https://doi.org/10.1098/rspb.2020.077

Li, Y., Liu, X., Li, X., Petitpierre, B., & Guisan, A. (2014). Residence time, expansion toward the equator in the invaded range and native range size matter to climatic niche shifts in non-native species. *Global Ecology and Biogeography, 23*(10), 1094–1104. https://doi.org/10.1111/geb.12191

Lindemann-Matthies, P. (2016). Beasts or beauties? Laypersons' perception of invasive alien plant species in Switzerland and attitudes towards their management. *NeoBiota, 29*, 15–33. https://doi.org/10.3897/neobiota.29.5786

Lindenmayer, D. B., Fischer, J., Felton, A., Crane, M., Michael, D., Macgregor, C., Montague-Drake, R., Manning, A., & Hobbs, R. J. (2008). Novel ecosystems resulting from landscape transformation create dilemmas for modern conservation practice. *Conservation Letters, 1*(3), 129–135. https://doi.org/10.1111/j.1755-263X.2008.00021.x

Lishawa, S. C., Lawrence, B. A., Albert, D. A., Larkin, D. J., & Tuchman, N. C. (2019). Invasive species removal increases species and phylogenetic diversity of wetland plant communities. *Ecology and Evolution, 9*(11), 6231–6244. https://doi.org/10.1002/ece3.5188

Liu, X., Blackburn, T. M., Song, T., Wang, X., Huang, C., & Li, Y. (2020). Animal invaders threaten protected areas worldwide. *Nature Communications, 11*(1), 2892. https://doi.org/10.1038/s41467-020-16719-2

Liu, X., Wang, S., Ke, Z., Cheng, C., Wang, Y., Zhang, F., Xu, F., Li, X., Gao, X., Jin, C., Zhu, W., Yan, S., & Li, Y. (2018). More invaders do not result in heavier impacts: The effects of non-native bullfrogs on native anurans are mitigated by high densities of non-native crayfish. *Journal of Animal Ecology, 87*(3), 850–862. https://doi.org/10.1111/1365-2656.12793

Lockwood, J. L., Cassey, P., & Blackburn, T. M. (2005). The role of propagule pressure in explaining species invasions. *Trends in Ecology & Evolution, 20*(5), 223–228. https://doi.org/10.1016/j.tree.2005.02.004

Lockwood, J. L., Cassey, P., & Blackburn, T. M. (2009). The more you introduce the more you get: The role of colonization pressure and propagule pressure in invasion ecology. *Diversity and Distributions, 15*(5), 904–910. https://doi.org/10.1111/j.1472-4642.2009.00594.x

Lockwood, J. L., Lieurance, D., Flory, S. L., Meyerson, L. A., Ricciardi, A., & Simberloff, D. (2023). Moving scholarship on invasion science forward. *Trends in Ecology & Evolution, 38*(6). https://doi.org/10.1016/j.tree.2023.01.006

Lombaert, E., Guillemaud, T., Cornuet, J.-M., Malausa, T., Facon, B., & Estoup, A. (2010). Bridgehead Effect in the Worldwide Invasion of the Biocontrol Harlequin Ladybird. *PLoS ONE, 5*(3), e9743. https://doi.org/10.1371/journal.pone.0009743

Lomolino, M. V., Riddle, B. R., Whittake, R. J., & Brown, J. H. (2010). *Biogeography* (Fourth Edition, Vol. 87). Sinauer Associates. https://www.journals.uchicago.edu/doi/10.1086/665420

Lord, J. P., Barry, J. P., & Graves, D. (2017). Impact of climate change on direct and indirect species interactions. *Marine Ecology Progress Series, 571*, 1–11. https://doi.org/10.3354/meps12148

Lugo, A. E. (2004). The outcome of alien tree invasions in Puerto Rico. *Frontiers in Ecology and the Environment, 2*(5), 265–273. https://doi.org/10.1890/1540-9295(2004)002[0265:TOOATI]2.0.CO;2

MacDougall, A. S., & Turkington, R. (2005). Are Invasive Species the Drivers or Passengers of Change in Degraded Ecosystems? *Ecology, 86*(1), 42–55. https://doi.org/10.1890/04-0669

Mace, G. M. (2014). Whose conservation? *Science, 345*(6204), 1558–1560. https://doi.org/10.1126/science.1254704

Maclean, K., Robinson, C., Bock, E., & Rist, P. (2022). Reconciling risk and responsibility on Indigenous country: Bridging the boundaries to guide knowledge sharing for cross-cultural biosecurity risk management in northern Australia. *Journal of Cultural Geography, 39*(1), 32–54. https://doi.org/10.1080/08873631.2021.1911078

MacPhail, V. J., & Colla, S. R. (2020). Power of the people: A review of citizen science programs for conservation. *Biological Conservation, 249*, 108739. https://doi.org/10.1016/j.biocon.2020.108739

Magellan, K. (2019). Prayer animal release: An understudied pathway for introduction of invasive aquatic species. *Aquatic Ecosystem Health & Management, 22*(4), 452–461. https://doi.org/10.1080/14634988.2019.1691433

Mankad, A., Kennedy, U., & Carter, L. (2019). Biological control of pests and a social model of animal welfare. *Journal of Environmental Management, 247*, 313–322. https://doi.org/10.1016/j.jenvman.2019.06.080

Martinez, B., Reaser, J. K., Dehgan, A., Zamft, B., Baisch, D., McCormick, C., Giordano, A. J., Aicher, R., & Selbe, S. (2020). Technology innovation: Advancing capacities for the early detection of and rapid response to invasive species. *Biological Invasions, 22*(1), 75–100. https://doi.org/10.1007/s10530-019-02146-y

Mascaro, J., Becklund, K. K., Hughes, R. F., & Schnitzer, S. A. (2008). Limited native plant regeneration in novel, exotic-dominated forests on Hawai'i. *Forest Ecology and Management, 256*(4), 593–606. https://doi.org/10.1016/j.foreco.2008.04.053

McGeoch, M. A., & Jetz, W. (2019). Measure and Reduce the Harm Caused by Biological Invasions. *One Earth, 1*(2), 171–174. https://doi.org/10.1016/j.oneear.2019.10.003

McKinney, M. L., & Lockwood, J. L. (1999). Biotic homogenization: A few winners replacing many losers in the next mass extinction. *Trends in Ecology & Evolution, 14*(11), 450–453. https://doi.org/10.1016/S0169-5347(99)01679-1

McNeely, J. A. (Ed.). (2001). *The great reshuffling: Human dimensions of invasive alien species.* IUCN. https://portals.iucn.org/library/node/7850

McWilliam, A. (2000). A Plague on Your House? Some Impacts of *Chromolaena odorata* on Timorese Livelihoods. *Human Ecology, 28*(3), 451–469. https://cir.nii.ac.jp/crid/1574231875722232192

Mellor, D. J., Beausoleil, N. J., Littlewood, K. E., McLean, A. N., McGreevy, P. D., Jones, B., & Wilkins, C. (2020). The 2020 Five Domains Model: Including Human–Animal Interactions in Assessments of Animal Welfare. *Animals, 10*(10), Article 10. https://doi.org/10.3390/ani10101870

Meyerson, F. A. B., Meyerson, L. A., & Reaser, J. K. (2009). Biosecurity from the ecologist's perspective: Developing a more comprehensive approach. *International Journal of Risk Assessment and Management, 12*(2/3/4), 147. https://www.academia.edu/20622951/Biosecurity_from_the_ecologists_perspective_developing_a_more_comprehensive_approach

Meyerson, L. A., & Reaser, J. K. (2002). Biosecurity: Moving toward a comprehensive approach: A comprehensive approach to biosecurity is necessary to minimize the risk of harm caused by non-native organisms to agriculture, the economy, the environment, and human health. *BioScience, 52*(7), 593–600. https://doi.org/10.1641/0006-3568(2002)052[0593:BMTACA]2.0.CO;2

Meyerson, L. A., & Reaser, J. K. (2003). Bioinvasions, bioterrorism, and biosecurity. *Frontiers in Ecology and the Environment, 1*(6), 307–314. https://doi.org/10.1890/1540-9295(2003)001[0307:BBAB]2.0.CO;2

Meyerson, L. A., Reaser, J. K., & Chyba, C. F. (2002). A unified definition of biosecurity. (Letters). *Science, 295*(5552), 44–45. https://go.gale.com/ps/i.do?p=HRCA&sw=w&issn=00368075&v=2.1&it=r&id=GALE%7CA82013232&sid=googleScholar&linkaccess=abs

Michán, L. (2011). Bibliometric analysis of systematics production in Latin America. *Acta Biologica Colombiana, 16*(2), 33–46. https://www.scopus.com/inward/record.uri?eid=2-s2.0-84860908473&partnerID=40&md5=df709314613bd69eb0969237676ca41e

Millennium Ecosystem Assessment. (2005). *Ecosystems and Human Well-Being: Opportunities and Challenges for Business and Industry* (Vol. 5). Island Press. https://www.unioviedo.es/ranadon/Ricardo_Anadon/docencia/DoctoradoEconomia/Millenium%20Eco%20Assesment%2005%20Oppor%20Business%20Industry.pdf

Miller, J. R., & Bestelmeyer, B. T. (2016). What's wrong with novel ecosystems, really? *Restoration Ecology, 24*(5), 577–582. https://doi.org/10.1111/rec.12378

Miller-Rushing, A., Primack, R., & Bonney, R. (2012). The history of public participation in ecological research. *Frontiers in Ecology and the Environment, 10*(6), 285–290. https://doi.org/10.1890/110278

Mooney, H. A., & Hobbs, R. J. (2000). Invasive Species in a Changing World. In *Land use change and invasive* (pp. 55–64). Ecological Restoration. https://www.jstor.org/stable/43440925

Mooney, H. A., Mack, R., McNeely, J. A., Neville, L. E., Schei, P. J., & Waage, J. K. (Eds.). (2005). *Invasive Alien Species: A New Synthesis* (Vol. 63). Island Press. https://books.google.co.za/books?id=KBqYU2dC-b4C

Moore, J. W. (2016). *Anthropocene or Capitalocene? Nature, History, and the Crisis of Capitalism* (p. 24). PM Press. https://orb.binghamton.edu/sociology_fac/1

Muller, S., Power, E. R., Suchet-Pearson, S., Wright, S., & Lloyd, K. (2009). "Quarantine Matters!": Quotidian Relationships around Quarantine in Australia's Northern Borderlands. *Environment and Planning A: Economy and Space, 41*(4), 780–795. https://doi.org/10.1068/a40196

Müller-Schärer, H., Bouchemousse, S., Litto, M., McEvoy, P. B., Roderick, G. K., & Sun, Y. (2020). How to better predict long-term benefits and risks in weed biocontrol: An evolutionary perspective. *Current Opinion in Insect Science, 38*, 84–91. https://doi.org/10.1016/j.cois.2020.02.006

Munishi, L. K., & Ngondya, I. B. (2022). Realizing UN decade on ecosystem restoration through a nature-based approach: A case review of management of biological invasions in protected areas. *PLoS Sustainability and Transformation, 1*(8), e0000027. https://doi.org/10.1371/journal.pstr.0000027

Murcia, C., Aronson, J., Kattan, G. H., Moreno-Mateos, D., Dixon, K., & Simberloff, D. (2014). A critique of the 'novel ecosystem' concept. *Trends in Ecology & Evolution, 29*(10), 548–553. https://doi.org/10.1016/j.tree.2014.07.006

Murphy, S. T., Subedi, N., Jnawali, S. R., Lamichhane, B. R., Upadhyay, G. P., Kock, R., & Amin, R. (2013). Invasive mikania in Chitwan National Park, Nepal: The threat to the greater one-horned rhinoceros *Rhinoceros unicornis* and factors driving the invasion. *Oryx, 47*(3), 361–368. https://doi.org/10.1017/S003060531200124X

Newbold, T., Hudson, L. N., Hill, S. L. L., Contu, S., Lysenko, I., Senior, R. A., Börger, L., Bennett, D. J., Choimes, A., Collen, B., Day, J., De Palma, A., Díaz, S., Echeverria-Londoño, S., Edgar, M. J., Feldman, A., Garon, M., Harrison, M. L. K., Alhusseini, T., … Purvis, A. (2015). Global effects of land use on local terrestrial biodiversity. *Nature, 520*(7545), 45–50. https://doi.org/10.1038/nature14324

Nixon, R. (2011). *Slow Violence and the Environmentalism of the Poor.* Harvard University Press.

Novoa, A., Dehnen-Schmutz, K., Fried, J., & Vimercati, G. (2017). Does public awareness increase support for invasive species management? Promising evidence across taxa and landscape types. *Biological Invasions, 19*(12), 3691–3705. https://doi.org/10.1007/s10530-017-1592-0

Nuñez, M. A., Davis, K. T., Dimarco, R. D., Peltzer, D. A., Paritsis, J., Maxwell, B. D., & Pauchard, A. (2021). Should tree invasions be used in treeless ecosystems to mitigate climate change? *Frontiers in Ecology and the Environment, 19*(6), 334–341. https://doi.org/10.1002/fee.2346

Nuñez, M. A., Pauchard, A., & Ricciardi, A. (2020). Invasion Science and the Global Spread of SARS-CoV-2. *Trends in Ecology & Evolution, 35*(8), 642–645. https://doi.org/10.1016/j.tree.2020.05.004

Occhipinti-Ambrogi, A., & Galil, B. S. (2004). A uniform terminology on bioinvasions: A chimera or an operative tool? *Marine Pollution Bulletin, 49*(9), 688–694. https://doi.org/10.1016/j.marpolbul.2004.08.011

Ogden, N. H., Wilson, J. R. U., Richardson, D. M., Hui, C., Davies, S. J., Kumschick, S., Le Roux, J. J., Measey, J., Saul, W.-C., & Pulliam, J. R. C. (2019). Emerging infectious diseases and biological invasions: A call for a One Health collaboration in science and management. *Royal Society Open Science, 6*(3), 181577. https://doi.org/10.1098/rsos.181577

Olden, J. D. (2006). Biotic homogenization: A new research agenda for conservation biogeography. *Journal of Biogeography, 33*(12), 2027–2039. https://doi.org/10.1111/j.1365-2699.2006.01572.x

OpIndia. (2021, February 10). As Priyanka Gandhi cites Nehru to attack the new Farm Laws, here's how India's first PM gifted poisonous 'Congress grass' to the nation. *OpIndia.* URL: https://www.opindia.com/2021/02/heres-how-indias-first-pm-gifted-poisonous-congress-grass-to-the-nation/

Parker, I. M., Simberloff, D., Lonsdale, W. M., Goodell, K., Wonham, M., Kareiva, P. M., Williamson, M. H., Von Holle, B., Moyle, P. B., Byers, J. E., & Goldwasser, L. (1999). Impact: Toward a Framework for Understanding the Ecological Effects of Invaders. *Biological Invasions, 1*(1), 3–19. https://doi.org/10.1023/A:1010034312781

Parravicini, V., Azzurro, E., Kulbicki, M., & Belmaker, J. (2015). Niche shift can impair the ability to predict invasion risk in the marine realm: An illustration using Mediterranean fish invaders. *Ecology Letters, 18*(3), 246–253. https://doi.org/10.1111/ele.12401

Pauchard, A., & Shea, K. (2006). Integrating the Study of Non-native Plant Invasions across Spatial Scales. *Biological Invasions, 8*(3), 399–413. https://doi.org/10.1007/s10530-005-6419-8

Pauly, D. (1995). Anecdotes and the shifting baseline syndrome of fisheries. *Trends in Ecology & Evolution, 10*(10), 430. https://doi.org/10.1016/S0169-5347(00)89171-5

Pawson, E., & Christensen, A. A. (2014). Landscapes of the Anthropocene: From dominion to dependence? In *Rethinking Invasion Ecologies from the Environmental Humanities* (pp. 82–102). Routledge.

Peltzer, D. A., Bellingham, P. J., Dickie, I. A., Houliston, G., Hulme, P. E., Lyver, P. O., McGlone, M., Richardson, S. J., & Wood, J. (2019). Scale and complexity implications of making New Zealand predator-free by 2050. *Journal of the Royal Society of New Zealand, 49*(3), 412–439. https://doi.org/10.1080/03036758.2019.1653940

Perry, D., & Perry, G. (2008). Improving interactions between animal rights groups and conservation biologists. *Conservation Biology, 22*(1), 27–35. https://doi.org/10.1111/j.1523-1739.2007.00845.x

Pfeiffer, J. M., & Voeks, R. A. (2008). Biological invasions and biocultural diversity: Linking ecological and cultural systems. *Environmental Conservation, 35*(4), 281–293. https://doi.org/10.1017/S0376892908005146

Pimm, S. L., Jenkins, C. N., Abell, R., Brooks, T. M., Gittleman, J. L., Joppa, L. N., Raven, P. H., Roberts, C. M., & Sexton, J. O. (2014). The biodiversity of species and their rates of extinction, distribution, and protection. *Science, 344*(6187), 1246752. https://doi.org/10.1126/science.1246752

Pino, J., Font, X., Carbó, J., Jové, M., & Pallarès, L. (2005). Large-scale correlates of alien plant invasion in Catalonia (NE of Spain). *Biological Conservation, 122*(2), 339–350. https://doi.org/10.1016/j.biocon.2004.08.006

Platenberg, R. J., Raymore, M., Primack, A., & Troutman, K. (2020). Monitoring Vocalizing Species by Engaging Community Volunteers Using Cell Phones. *Wildlife Society Bulletin, 44*(4), 782–789. https://doi.org/10.1002/wsb.1141

Pocock, M. J. O., Chandler, M., Bonney, R., Thornhill, I., Albin, A., August, T., Bachman, S., Brown, P. M. J., Cunha, D. G. F., Grez, A., Jackson, C., Peters, M., Rabarijaon, N. R., Roy, H. E., Zaviezo, T., & Danielsen, F. (2018). A Vision for Global Biodiversity Monitoring With Citizen Science. In *Advances in Ecological Research* (Vol. 59, pp. 169–223). Elsevier. https://doi.org/10.1016/bs.aecr.2018.06.003

Pocock, M. J. O., Chapman, D. S., Sheppard, L. J., & Roy, H. E. (2014). *Choosing and using citizen science: A guide to when and how to use citizen science to monitor biodiversity and the environment.* https://nora.nerc.ac.uk/id/eprint/510644/1/N510644CR.pdf

Pocock, M. J. O., Roy, H. E., Preston, C. D., & Roy, D. B. (2015). The Biological Records Centre: A pioneer of citizen science. *Biological Journal of the Linnean Society, 115*(3), 475–493. https://doi.org/10.1111/bij.12548

Pollock, L. J., O'Connor, L. M. J., Mokany, K., Rosauer, D. F., Talluto, M. V., & Thuiller, W. (2020). Protecting Biodiversity (in All Its Complexity): New Models and Methods. *Trends in Ecology & Evolution, 35*(12), 1119–1128. https://doi.org/10.1016/j.tree.2020.08.015

Pörtner, H.-O., Scholes, R. J., Agard, J., Archer, E., Bai, X., Barnes, D., Burrows, M., Chan, L., Cheung, W. L. (William), Diamond, S., Donatti, C., Duarte, C., Eisenhauer, N., Foden, W., Gasalla, M. A., Handa, C., Hickler, T., Hoegh-Guldberg, O., Ichii, K., … Ngo, H. (2021). *IPBES-IPCC co-sponsored workshop report on biodiversity and climate change* (Version 2). Zenodo. https://doi.org/10.5281/ZENODO.4782538

Powell, M. C., & Colin, M. (2009). Participatory paradoxes: Facilitating citizen engagement in science and technology from the Top-Down? *Bulletin of Science, Technology & Society, 29*(4), 325–342. https://doi.org/10.1177/0270467609336308

Puri, R. K. (2015). The uniqueness of the everyday: Herders and invasive species in India. In J. Barnes & M. R. Dove (Eds.), *Climate Cultures: Anthropological Perspectives on Climate Change* (pp. 249–272). Yale University Press. http://yalebooks.com/book/9780300198812/climate-cultures

Pyke, D. A., Chambers, J. C., Beck, J. L., Brooks, M. L., & Mealor, B. A. (2016). Land Uses, Fire, and Invasion: Exotic Annual *Bromus* and Human Dimensions. In M. J. Germino, J. C. Chambers, & C. S. Brown (Eds.), *Exotic Brome-Grasses in Arid and Semiarid Ecosystems of the Western US* (pp. 307–337). Springer International Publishing. https://doi.org/10.1007/978-3-319-24930-8_11

Pyšek, P., Bacher, S., Kühn, I., Novoa, A., Catford, J. A., Hulme, P. E., Pergl, J., Richardson, D. M., Wilson, J. R. U., & Blackburn, T. M. (2020). MAcroecological Framework for Invasive Aliens (MAFIA): Disentangling large-scale context dependence in biological invasions. *NeoBiota, 62*, 407–461. https://doi.org/10.3897/neobiota.62.52787

Pyšek, P., Blackburn, T. M., García-Berthou, E., Perglová, I., & Rabitsch, W. (2017). Displacement and Local Extinction of Native and Endemic Species. In M. Vilà & P. E. Hulme (Eds.), *Impact of Biological Invasions on Ecosystem Services* (Vol. 12, pp. 157–175). Springer International Publishing. https://doi.org/10.1007/978-3-319-45121-3_10

Pyšek, P., Hulme, P. E., Simberloff, D., Bacher, S., Blackburn, T. M., Carlton, J. T., Dawson, W., Essl, F., Foxcroft, L. C., Genovesi, P., Jeschke, J. M., Kühn, I., Liebhold, A. M., Mandrak, N. E., Meyerson, L. A., Pauchard, A., Pergl, J., Roy, H. E., Seebens, H., … Richardson, D. M. (2020). Scientists' warning on invasive alien species. *Biological Reviews, 95*(6), 1511–1534. https://doi.org/10.1111/brv.12627

Pyšek, P., Jarošík, V., & Pergl, J. (2011). Alien plants introduced by different pathways differ in invasion success: Unintentional introductions as a threat to natural areas. *PLoS ONE, 6*(9), e24890. https://doi.org/10.1371/journal.pone.0024890

Pyšek, P., Pergl, J., Essl, F., Lenzner, B., Dawson, W., Kreft, H., Weigelt, P., Winter, M., Kartesz, J., Nishino, M., Antonova, L. A., Barcelona, J. F., Cabesaz, F. J., Cárdenas, D., Cárdenas-Toro, J., Castaño, N., Chacón, E., Chatelain, C., Dullinger, S., … Kleunen, M. van. (2017). Naturalized alien flora of the world: Species diversity, taxonomic and phylogenetic patterns, geographic distribution and global hotspots of plant invasion. *Preslia, 89*(3), 203–274. https://doi.org/10.23855/preslia.2017.203

Pyšek, P., & Richardson, D. M. (2010). Invasive species, environmental change and management, and health. *Annual Review of Environment and Resources, 35*(1), 25–55. https://doi.org/10.1146/annurev-environ-033009-095548

Pyšek, P., Richardson, D. M., Rejmánek, M., Webster, G. L., Williamson, M., & Kirschner, J. (2004). Alien plants in checklists and floras: Towards better communication between taxonomists and ecologists. *TAXON, 53*(1), 131–143. https://doi.org/10.2307/4135498

Qvenild, M., Setten, G., & Skår, M. (2014). Politicising plants: Dwelling and invasive alien species in domestic gardens in Norway. *Norsk Geografisk Tidsskrift – Norwegian Journal of Geography, 68*(1), 22–33. https://doi.org/10.1080/00291951.2013.870599

Rai, R. K., & Scarborough, H. (2015). Understanding the Effects of the Invasive Plants on Rural Forest-dependent Communities. *Small-Scale Forestry, 14*(1), 59–72. https://doi.org/10.1007/s11842-014-9273-7

Ramakrishnan, P. S., & Vitousek, P. M. (1989). Ecosystem-level processes and the consequences of biological invasions. In *Biological Invasions. A Global Perspective* (pp. 281–300). Wiley.

Rangan, H., & Kull, C. A. (2009). What makes ecology `political'?: Rethinking `scale' in political ecology. *Progress in Human Geography, 33*(1), 28–45. https://doi.org/10.1177/0309132508090215

Reaser, J. K., Simpson, A., Guala, G. F., Morisette, J. T., & Fuller, P. (2020). Envisioning a national invasive species information framework. *Biological Invasions, 22*(1), 21–36. https://doi.org/10.1007/s10530-019-02141-3

Redding, D. W., Pigot, A. L., Dyer, E. E., Şekercioğlu, Ç. H., Kark, S., & Blackburn, T. M. (2019). Location-level processes drive the establishment of alien bird populations worldwide. *Nature, 571*(7763), 103–106. https://doi.org/10.1038/s41586-019-1292-2

Redford, K. H., Brooks, T. M., Macfarlane, N. B. W., & Adams, J. S. (Eds.). (2019). *Genetic frontiers for conservation: An assessment of synthetic biology and biodiversity conservation: technical assessment.* IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2019.05.en

Rejmanek, M., & Richardson, D. M. (1996). What Attributes Make Some Plant Species More Invasive? *Ecology, 77*(6), 1655–1661. https://doi.org/10.2307/2265768

Reo, N. J., & Ogden, L. A. (2018). Anishnaabe Aki: An indigenous perspective on the global threat of invasive species. *Sustainability Science, 13*(5), 1443–1452. https://doi.org/10.1007/s11625-018-0571-4

Rettberg, S. (2010). Contested narratives of pastoral vulnerability and risk in Ethiopia's Afar region. *Pastoralism – Research, Policy and Practice, 1*(2), 248–273. https://doi.org/10.3362/2041-7136.2010.014

Ricciardi, A., Blackburn, T. M., Carlton, J. T., Dick, J. T. A., Hulme, P. E., Iacarella, J. C., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Sutherland, W. J., Wardle, D. A., & Aldridge, D. C. (2017). Invasion Science: A Horizon Scan of Emerging Challenges and Opportunities. *Trends in Ecology & Evolution, 32*(6), 464–474. https://doi.org/10.1016/j.tree.2017.03.007

Ricciardi, A., Hoopes, M. F., Marchetti, M. P., & Lockwood, J. L. (2013). Progress toward understanding the ecological impacts of nonnative species. *Ecological Monographs, 83*(3), 263–282. https://doi.org/10.1890/13-0183.1

Richardson, D. M. (1998). Forestry Trees as Invasive Aliens. *Conservation Biology, 12*(1), 18–26. https://doi.org/10.1046/j.1523-1739.1998.96392.x

Richardson, D. M., Pyšek, P., & Carlton, J. T. (2010). A Compendium of Essential Concepts and Terminology in Invasion Ecology. In D. M. Richardson (Ed.), *Fifty Years of Invasion Ecology* (pp. 409–420). Wiley-Blackwell. https://doi.org/10.1002/9781444329988.ch30

Richter, R., Berger, U. E., Dullinger, S., Essl, F., Leitner, M., Smith, M., & Vogl, G. (2013). Spread of invasive ragweed: Climate change, management and how to reduce allergy costs. *Journal of Applied Ecology, 50*(6), 1422–1430. https://doi.org/10.1111/1365-2664.12156

Ritchie, H., Mathieu, E., Rodés-Guirao, L., Appel, C., Giattino, C., Ortiz-Ospina, E., Hasell, J., Macdonald, B., Beltekian, D., & Roser, M. (2020). Coronavirus Pandemic (COVID-19). *Our World in Data.* https://ourworldindata.org/mortality-risk-covid

Ritvo, H. (2014). *How Wild is Wild?* [Application/pdf]. 7 Pages. https://doi.org/10.5282/RCC/6270

Robbins, L. E. (2002). *Elephant slaves and pampered parrots: Exotic animals in eighteenth-century Paris.* JHU Press.

Robinson, T. B., Martin, N., Loureiro, T. G., Matikinca, P., & Robertson, M. P. (2020). Double trouble: The implications of climate change for biological invasions. *NeoBiota, 62*, 463–487. https://doi.org/10.3897/neobiota.62.55729

Rodda, G. H., & Savidge, J. A. (2007). Biology and Impacts of Pacific Island Invasive Species. 2. *Boiga irregularis*, the Brown Tree Snake (Reptilia: Colubridae) 1. *Pacific Science, 61*(3), 307–324. https://doi.org/10.2984/1534-6188(2007)61[307:BAIOPI]2.0.CO;2

Rojas-Sandoval, J., & Acevedo-Rodríguez, P. (2015). Naturalization and invasion of alien plants in Puerto Rico and the Virgin Islands. *Biological Invasions, 17*(1), 149–163. https://doi.org/10.1007/s10530-014-0712-3

Roman, J., & Darling, J. A. (2007). Paradox lost: Genetic diversity and the success of aquatic invasions. *Trends in Ecology & Evolution, 22*(9), 454–464. https://doi.org/10.1016/j.tree.2007.07.002

Ross, D. J., Johnson, C. R., Hewitt, C. L., & Ruiz, G. M. (2004). Interaction and impacts of two introduced species on a soft-sediment marine assemblage in SE Tasmania. *Marine Biology, 144*(4), 747–756. https://doi.org/10.1007/s00227-003-1223-4

Rotherham, I. D., & Lambert, R. A. (Eds.). (2013). *Invasive and introduced plants and animals: Human perceptions, attitudes and approaches to management.* Routledge. https://www.crcpress.com/Invasive-and-Introduced-Plants-and-Animals-Human-Perceptions-Attitudes/Rotherham-Lambert/p/book/9780415830690

Roura-Pascual, N., Leung, B., Rabitsch, W., Rutting, L., Vervoort, J., Bacher, S., Dullinger, S., Erb, K.-H., Jeschke, J. M., Katsanevakis, S., Kühn, I., Lenzner, B., Liebhold, A. M., Obersteiner, M., Pauchard, A., Peterson, G. D., Roy, H. E., Seebens, H., Winter, M., … Essl, F. (2021). Alternative futures for global biological invasions. *Sustainability Science, 16*(5), 1637–1650. https://doi.org/10.1007/s11625-021-00963-6

Roy, H. E., Hesketh, H., Purse, B. V., Eilenberg, J., Santini, A., Scalera, R., Stentiford, G. D., Adriaens, T., Bacela-Spychalska, K., Bass, D., Beckmann, K. M., Bessell, P., Bojko, J., Booy, O., Cardoso, A. C., Essl, F., Groom, Q., Harrower, C., Kleespies, R., … Dunn, A. M. (2017). Alien Pathogens on the Horizon: Opportunities for Predicting their Threat to Wildlife. *Conservation Letters, 10*(4), 477–484. https://doi.org/10.1111/conl.12297

Roy, H. E., & Lawson Handley, L.-J. (2012). Networking: A community approach to invaders and their parasites. *Functional Ecology, 26*(6), 1238–1248. https://doi.org/10.1111/j.1365-2435.2012.02032.x

Roy, H. E., Peyton, J. M., & Booy, O. (2020). Guiding principles for utilizing social influence within expert-elicitation to inform conservation decision-making. *Global Change Biology, 26*(6), 3181–3184. https://doi.org/10.1111/gcb.15062

Roy, H. E., Rorke, S. L., Beckmann, B., Booy, O., Botham, M. S., Brown, P. M. J., Harrower, C., Noble, D., Sewell, J., & Walker, K. (2015). The contribution of volunteer recorders to our understanding of biological invasions. *Biological Journal* *of the Linnean Society, 115*(3), 678–689. https://doi.org/10.1111/bij.12518

Roy, H. E., Tricarico, E., Hassall, R., Johns, C. A., Roy, K. A., Scalera, R., Smith, K. G., & Purse, B. V. (2023). The role of invasive alien species in the emergence and spread of zoonoses. *Biological Invasions, 25*(4), 1249–1264. https://doi.org/10.1007/s10530-022-02978-1

Russell, J. C., Meyer, J.-Y., Holmes, N. D., & Pagad, S. (2017). Invasive alien species on islands: Impacts, distribution, interactions and management. *Environmental Conservation, 44*(4), 359–370. https://doi.org/10.1017/S0376892917000297

Sagarin, R., & Pauchard, A. (2012). *Observation and Ecology*. Island Press/Center for Resource Economics. https://doi.org/10.5822/978-1-61091-230-3

Sala, E., Kizilkaya, Z., Yildirim, D., & Ballesteros, E. (2011). Alien Marine Fishes Deplete Algal Biomass in the Eastern Mediterranean. *PLoS ONE, 6*(2), e17356. https://doi.org/10.1371/journal.pone.0017356

Sala, O. E., Chapin, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L. F., Jackson, R. B., Kinzig, A., Leemans, R., Lodge, D. M., Mooney, H. A., Oesterheld, M., Poff, N. L., Sykes, M. T., Walker, B. H., Walker, M., & Wall, D. H. (2000). Biodiversity—Global biodiversity scenarios for the year 2100. *Science, 287*(5459), 1770–1774. https://doi.org/10.1126/science.287.5459.1770

Salmón, E. (2000). Kincentric Ecology: Indigenous Perceptions of the Human-Nature Relationship. *Ecological Applications, 10*(5), 1327–1332. https://doi.org/10.2307/2641288

Sapsford, S. J., Brandt, A. J., Davis, K. T., Peralta, G., Dickie, I. A., Gibson, R. D., Green, J. L., Hulme, P. E., Nuñez, M. A., Orwin, K. H., Pauchard, A., Wardle, D. A., & Peltzer, D. A. (2020). Towards a framework for understanding the context dependence of impacts of non-native tree species. *Functional Ecology, 34*(5), 944–955. https://doi.org/10.1111/1365-2435.13544

Saul, W.-C., & Jeschke, J. M. (2015). Eco-evolutionary experience in novel species interactions. *Ecology Letters, 18*(3), 236–245. https://doi.org/10.1111/ele.12408

Saul, W.-C., Roy, H. E., Booy, O., Carnevali, L., Chen, H.-J., Genovesi, P., Harrower, C. A., Hulme, P. E., Pagad, S., Pergl, J., & Jeschke, J. M. (2017). Assessing patterns in introduction pathways of alien species by linking major invasion data bases. *Journal of Applied Ecology, 54*(2), 657–669. https://doi.org/10.1111/1365-2664.12819

Scanlon, E., Woods, W., & Clow, D. (2014). Informal participation in science in the UK: identification, location and mobility with iSpot. *Journal of Educational Technology & Society, 17*(2), 58–71.

Schelhas, J., Alexander, J., Brunson, M., Cabe, T., Crall, A., Dockry, M. J., Emery, M. R., Frankel, S. J., Hapner, N., Hickman, C. R., Jordan, R., LaVoie, M. J., Ma, Z., Starinchak, J., & Vukomanovic, J. (2021). Social and Cultural Dynamics of Non-native Invasive Species. In T. M. Poland, T. Patel-Weynand, D. M. Finch, C. F. Miniat, D. C. Hayes, & V. M. Lopez (Eds.), *Invasive Species in Forests and Rangelands of the United States* (pp. 267–291). Springer International Publishing. https://doi.org/10.1007/978-3-030-45367-1_12

Schweiger, O., Biesmeijer, J. C., Bommarco, R., Hickler, T., Hulme, P. E., Klotz, S., Kühn, I., Moora, M., Nielsen, A., Ohlemüller, R., Petanidou, T., Potts, S. G., Pyšek, P., Stout, J. C., Sykes, M. T., Tscheulin, T., Vilà, M., Walther, G.-R., Westphal, C., … Settele, J. (2010). Multiple stressors on biotic interactions: How climate change and alien species interact to affect pollination. *Biological Reviews*, no-no. https://doi.org/10.1111/j.1469-185X.2010.00125.x

Secretariat of the CBD. (2020). *Global Biodiversity Outlook 5 -Summary for Policymakers*. https://www.cbd.int/gbo/gbo5/publication/gbo-5-spm-en.pdf

Seddon, N., Smith, A., Smith, P., Key, I., Chausson, A., Girardin, C., House, J., Srivastava, S., & Turner, B. (2021). Getting the message right on nature-based solutions to climate change. *Global Change Biology, 27*(8), 1518–1546. https://doi.org/10.1111/gcb.15513

Seebens, H. (2021). *SInAS database of alien species occurrences* (2.4.1) [dataset]. Zenodo. https://doi.org/10.5281/ZENODO.5562892

Seebens, H., Bacher, S., Blackburn, T. M., Capinha, C., Dawson, W., Dullinger, S., Genovesi, P., Hulme, P. E., Kleunen, M., Kühn, I., Jeschke, J. M., Lenzner, B., Liebhold, A. M., Pattison, Z., Pergl, J., Pyšek, P., Winter, M., & Essl, F. (2021). Projecting the continental accumulation of alien species through to 2050. *Global Change Biology, 27*(5), 970–982. https://doi.org/10.1111/gcb.15333

Seebens, H., Clarke, D. A., Groom, Q., Wilson, J. R. U., García-Berthou, E., Kühn, I., Roigé, M., Pagad, S., Essl, F., Vicente, J., Winter, M., & McGeoch, M. A. (2020). A workflow for standardising and integrating alien species distribution data. *NeoBiota, 59*, 39–59. https://doi.org/10.3897/neobiota.59.53578

Seebens, H., Essl, F., & Blasius, B. (2017). The intermediate distance hypothesis of biological invasions. *Ecology Letters, 20*(2), 158–165. https://doi.org/10.1111/ele.12715

Seebens, H., Essl, F., Dawson, W., Fuentes, N., Moser, D., Pergl, J., Pyšek, P., Van Kleunen, M., Weber, E., Winter, M., & Blasius, B. (2015). Global trade will accelerate plant invasions in emerging economies under climate change. *Global Change Biology, 21*(11), 4128–4140. https://doi.org/10.1111/gcb.13021

Selge, S., Fischer, A., & Van Der Wal, R. (2011). Public and professional views on invasive non-native species – A qualitative social scientific investigation. *Biological Conservation, 144*(12), 3089–3097. https://doi.org/10.1016/j.biocon.2011.09.014

Shackelford, N., Hobbs, R. J., Heller, N. E., Hallett, L. M., & Seastedt, T. R. (2013). Finding a middle-ground: The native/non-native debate. *Biological Conservation, 158*, 55–62. https://doi.org/10.1016/j.biocon.2012.08.020

Shackleton, R. T., Larson, B. M. H., Novoa, A., Richardson, D. M., & Kull, C. A. (2019). The human and social dimensions of invasion science and management. *Journal of Environmental Management, 229*, 1–9. https://doi.org/10.1016/j.jenvman.2018.08.041

Shackleton, R. T., Richardson, D. M., Shackleton, C. M., Bennett, B., Crowley, S. L., Dehnen-Schmutz, K., Estévez, R. A., Fischer, A., Kueffer, C., Kull, C. A., Marchante, E., Novoa, A., Potgieter, L. J., Vaas, J., Vaz, A. S., & Larson, B. M. H. (2019). Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management, 229*, 10–26. https://doi.org/10.1016/j.jenvman.2018.04.045

Shackleton, R. T., Shackleton, C. M., & Kull, C. A. (2019). The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management, 229*, 145–157. https://doi.org/10.1016/j.jenvman.2018.05.007

Shah, S. (2020). *The next great migration: The story of movement on a changing planet.* Bloomsbury Publishing.

Shea, K., & Chesson, P. (2002). Community ecology theory as a framework for biological invasions. *Trends in Ecology & Evolution, 17*(4), Article 4. https://doi.org/10.1016/S0169-5347(02)02495-3

Shove, E. (2010). Beyond the ABC: Climate Change Policy and Theories of Social Change. *Environment and Planning A: Economy and Space, 42*(6), 1273–1285. https://doi.org/10.1068/a42282

Shrestha, S. (2021). Impact of invasive alien species and gender. International *Journal of Research -GRANTHAALAYAH, 9*(1), 78–84. https://doi.org/10.29121/granthaalayah.v9.i1.2021.2859

Silvertown, J. (2009). A new dawn for citizen science. *Trends in Ecology & Evolution, 24*(9), 467–471. https://doi.org/10.1016/j.tree.2009.03.01

Simberloff, D. (2006). Invasional meltdown 6 years later: Important phenomenon, unfortunate metaphor, or both? *Ecology Letters, 9*(8), 912–919. https://doi.org/10.1111/j.1461-0248.2006.00939.x

Simberloff, D. (2011). How common are invasion-induced ecosystem impacts? *Biological Invasions, 13*(5), 1255–1268. https://doi.org/10.1007/s10530-011-9956-3

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vilà, M. (2013). Impacts of biological invasions: What's what and the way forward. *Trends in Ecology & Evolution, 28*(1), 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Simberloff, D., & Rejmanek, M. (Eds.). (2011). *Encyclopedia of Biological Invasions* (Vol. 3). University of California Press. https://doi.org/10.1525/9780520948433

Simberloff, D., & Von Holle, B. (1999). Positive Interactions of Nonindigenous Species: Invasional Meltdown? *Biological Invasions, 1*(1), 21–32. https://doi.org/10.1023/A:1010086329619

Sinclair, J. S., Brown, J. A., & Lockwood, J. L. (2020). Reciprocal human-natural system feedback loops within the invasion process. *NeoBiota, 62*, 489–508. https://doi.org/10.3897/neobiota.62.52664

Slimak, M. W., & Dietz, T. (2006). Personal Values, Beliefs, and Ecological Risk Perception. *Risk Analysis, 26*(6), 1689–1705. https://doi.org/10.1111/j.1539-6924.2006.00832.x

Smout, T. C. (Ed.). (2003). *People and Woods in Scotland: A History.* Edinburgh University Press. https://www.jstor.org/stable/10.3366/j.ctvxcrr4m

Steven, R., Barnes, M., Garnett, S. T., Garrard, G., O'Connor, J., Oliver, J. L., Robinson, C., Tulloch, A., & Fuller, R. A. (2019). Aligning citizen science with best practice: Threatened species conservation in Australia. *Conservation Science and Practice, 1*(10). https://doi.org/10.1111/csp2.100

Stoett, P. (2007). Counter-bioinvasion: Conceptual and governance challenges. *Environmental Politics, 16*(3), 433–452. https://doi.org/10.1080/09644010701251672

Stoett, P. (2010). Framing Bioinvasion: Biodiversity, Climate Change, Security, Trade, and Global Governance. *Global Governance: A Review of Multilateralism and International Organizations, 16*(1), 103–120. https://doi.org/10.1163/19426720-01601007

Stoett, P., Roy, H. E., & Pauchard, A. (2019). Invasive alien species and planetary and global health policy. *The Lancet Planetary Health, 3*(10), e400–e401. https://doi.org/10.1016/S2542-5196(19)30194-9

Stouffer, D. B., Cirtwill, A. R., & Bascompte, J. (2014). How exotic plants integrate into pollination networks. *Journal of Ecology, 102*(6), 1442–1450. https://doi.org/10.1111/1365-2745.12310

Sullivan, B. L., Aycrigg, J. L., Barry, J. H., Bonney, R. E., Bruns, N., Cooper, C. B., Damoulas, T., Dhondt, A. A., Dietterich, T., Farnsworth, A., Fink, D., Fitzpatrick, J. W., Fredericks, T., Gerbracht, J., Gomes, C., Hochachka, W. M., Iliff, M. J., Lagoze, C., La Sorte, F. A., … Kelling, S. (2014). The eBird enterprise: An integrated approach to development and application of citizen science. *Biological Conservation, 169*, 31–40. https://doi.org/10.1016/j.biocon.2013.11.003

Sunny, A., Diwakar, S., & Sharma, G. P. (2015). Native insects and invasive plants encounters. *Arthropod-Plant Interactions, 9*(4), 323–331. https://doi.org/10.1007/s11829-015-9384-x

Tassin, J., & Kull, C. A. (2015). Facing the broader dimensions of biological invasions. *Land Use Policy, 42*, 165–169. https://doi.org/10.1016/j.landusepol.2014.07.014

Taylor, K. T., Maxwell, B. D., McWethy, D. B., Pauchard, A., Nuñez, M. A., & Whitlock, C. (2017). *Pinus contorta* invasions increase wildfire fuel loads and may create a positive feedback with fire. *Ecology, 98*(3), 678–687. https://doi.org/10.1002/ecy.1673

Teacher, A. G. F., Griffiths, D. J., Hodgson, D. J., & Inger, R. (2013). Smartphones in ecology and evolution: A guide for the app-rehensive. *Ecology and Evolution, 3*(16), 5268–5278. https://doi.org/10.1002/ece3.888

Terefe, B., Williams, F., & Lamontagne-Godwin, J. (2020). *Invasive species management—Integrating a gender perspective.* CABI. https://doi.org/10.1079/CABICOMM-62-8140

Termeer, C. J. A. M., Dewulf, A., & Biesbroek, R. (2019). A critical assessment of the wicked problem concept: Relevance and usefulness for policy science and practice. *Policy and Society, 38*(2), 167–179. https://doi.org/10.1080/14494035.2019.1617971

Terry, J. C. D., Roy, H. E., & August, T. A. (2020). Thinking like a naturalist: Enhancing computer vision of citizen science images by harnessing contextual data. *Methods in Ecology and Evolution, 11*(2), 303–315. https://doi.org/10.1111/2041-210X.13335

Tessema, Y. A. (2012). Ecological and Economic Dimensions of the Paradoxical Invasive Species– *Prosopis juliflora* and Policy Challenges in Ethiopia. *Journal of Economics and Sustainable Development, 3*(8), 62. https://www.iiste.org/Journals/index.php/JEDS/article/view/2307

The Economist. (2022, October). *The pandemic's true death toll.* The Economist. https://www.economist.com/graphic-detail/coronavirus-excess-deaths-estimates

Thomas, C. D. (Ed.). (2017). *Inheritors of the Earth: How nature is thriving in an age of extinction.* Hachette UK. https://books.google.co.jp/books?id=eC7eDQAAQBAJ&printsec=frontcover&source=gbs_ge_summary_r&cad=0#v=onepage&q&f=false

Thomas, K. (1984). *Man and the Natural World Harmondsworth.* Penguin Books.

Thomas, M. B., & Willis, A. J. (1998). Biocontrol—Risky but necessary? *Trends in Ecology & Evolution, 13*(8), 325–329. https://doi.org/10.1016/S0169-5347(98)01417-7

Thornton, T. F., Puri, R. K., Bhagwat, S., & Howard, P. (2019). Human adaptation to biodiversity change: An adaptation process approach applied to a case study from southern India. *Ambio, 48*(12), 1431–1446. https://doi.org/10.1007/s13280-019-01225-7

Thresher, A. C. (2020). When Extinction Is Warranted: Invasive Species, Suppression-Drives and the Worst-Case Scenario. *Ethics, Policy & Environment, 25*(2), 132–152. https://doi.org/10.1080/21550085.2020.1848197

Trauernicht, C., Murphy, B. P., Tangalin, N., & Bowman, D. M. J. S. (2013). Cultural legacies, fire ecology, and environmental change in the Stone Country of Arnhem Land and Kakadu National Park, Australia. *Ecology and Evolution, 3*(2), 286–297. https://doi.org/10.1002/ece3.460

Traveset, A., Chamorro, S., Olesen, J. M., & Heleno, R. (2015). Space, time and aliens: Charting the dynamic structure of Galápagos pollination networks. *AoB PLANTS, 7*, plv068. https://doi.org/10.1093/aobpla/plv068

Traveset, A., & Richardson, D. M. (2014). Mutualistic Interactions and Biological Invasions. *Annual Review of Ecology, Evolution, and Systematics, 45*(1), 89–113. https://doi.org/10.1146/annurev-ecolsys-120213-091857

Trigger, D. S. (2008). Indigeneity, ferality, and what 'belongs' in the Australian bush: Aboriginal responses to 'introduced' animals and plants in a settler-descendant society. *Journal of the Royal Anthropological Institute, 14*(3), 628–646. https://doi.org/10.1111/j.1467-9655.2008.00521.x

Turbelin, A. J., Malamud, B. D., & Francis, R. A. (2017). Mapping the global state of invasive alien species: Patterns of invasion and policy responses. *Global Ecology and Biogeography, 26*(1), 78–92. https://doi.org/10.1111/geb.12517

Turner, B., Devisscher, T., Chabaneix, N., Woroniecki, S., Messier, C., & Seddon, N. (2022). The Role of Nature-Based Solutions in Supporting Social-Ecological Resilience for Climate Change Adaptation. *Annual Review of Environment and Resources, 47*(1), 123–148. https://doi.org/10.1146/annurev-environ-012220-010017

Udo, N., Darrot, C., & Atlan, A. (2019). From useful to invasive, the status of gorse on Reunion Island. *Journal of Environmental Management, 229*, 166–173. https://doi.org/10.1016/j.jenvman.2018.06.036

UNEP. (2021). *Joint tripartite and UNEP statement on definition of "One Health."* United Nations Environment Programme. http://www.unep.org/news-and-stories/statements/joint-tripartite-and-unep-statement-definition-one-health

UNEP, CBD, & WHO. (2015). *Connecting global priorities: Biodiversity and human health*. https://www.who.int/publications/i/item/9789241508537

Unger, S., Rollins, M., Tietz, A., & Dumais, H. (2021). iNaturalist as an engaging tool for identifying organisms in outdoor activities. *Journal of Biological Education, 55*(5), 537–547. https://doi.org/10.1080/00219266.2020.1739114

United Nations. (1992). *8. Convention on Biological Diversity*. https://treaties.un.org/pages/ViewDetails.aspx?src=TREATY&mtdsg_no=XXVII-8&chapter=27

United Nations. (2020a). *Goal 15 | Department of Economic and Social Affairs*. https://sdgs.un.org/goals/goal15

United Nations. (2020b). Take Action for the Sustainable Development Goals. *United Nations Sustainable Development.* https://www.un.org/sustainabledevelopment/sustainable-development-goals/

Vaarzon-Morel, P. (2010). Changes in Aboriginal perceptions of feral camels and of their impacts and management. *The Rangeland Journal, 32*(1), 73. https://doi.org/10.1071/RJ09055

van Kleunen, M., Dawson, W., Essl, F., Pergl, J., Winter, M., Weber, E., Kreft, H., Weigelt, P., Kartesz, J., Nishino, M., Antonova, L. A., Barcelona, J. F., Cabezas, F. J., Cárdenas, D., Cárdenas-Toro, J., Castaño, N., Chacón, E., Chatelain, C., Ebel, A. L., … Pyšek, P. (2015). Global exchange and accumulation of non-native plants. *Nature, 525*(7567), 100–103. https://doi.org/10.1038/nature14910

van Kleunen, M., Essl, F., Pergl, J., Brundu, G., Carboni, M., Dullinger, S., Early, R., González-Moreno, P., Groom, Q. J., Hulme, P. E., Kueffer, C., Kühn, I., Máguas, C., Maurel, N., Novoa, A., Parepa, M., Pyšek, P., Seebens, H., Tanner, R., … Dehnen-Schmutz, K. (2018). The changing role of ornamental horticulture in alien plant invasions. *Biological Reviews, 93*(3), 1421–1437. https://doi.org/10.1111/brv.12402

Van Wilgen, B. W., Dyer, C., Hoffmann, J. H., Ivey, P., Le Maitre, D. C., Moore, J. L., Richardson, D. M., Rouget, M., Wannenburgh, A., & Wilson, J. R. U. (2011). National-scale strategic approaches for managing introduced plants: Insights from Australian acacias in South Africa: Strategic approaches for managing introduced acacias. *Diversity and Distributions, 17*(5), 1060–1075. https://doi.org/10.1111/j.1472-4642.2011.00785.x

Vanbergen, A. J., & Initiative, T. I. P. (2013). Threats to an ecosystem service: Pressures on pollinators. *Frontiers in Ecology and the Environment, 11*(5), 251–259. https://doi.org/10.1890/120126

Vaz, A. S., Kueffer, C., Kull, C. A., Richardson, D. M., Schindler, S., Muñoz-Pajares, A. J., Vicente, J. R., Martins, J., Hui, C., Kühn, I., & Honrado, J. P. (2017). The progress of interdisciplinarity in invasion science. *Ambio, 46*(4), 428–442. https://doi.org/10.1007/s13280-017-0897-7

Vera, F. (2010). The shifting baseline syndrome in restoration ecology. In *Restoration and history* (1st ed., pp. 116–128). Routledge.

Verbrugge, L. N. H., Leuven, R. S. E. W., van Valkenburg, J. L. C. H., & van den Born, R. J. G. (2014). Evaluating stakeholder awareness and involvement in risk prevention of aquatic invasive plant species by a national code of conduct. *Aquatic Invasions, 9*(3), 369–381. https://doi.org/10.3391/ai.2014.9.3.11

Verbrugge, L. N. H., Leuven, R. S. E. W., & Zwart, Hub. A. E. (2016). Metaphors in Invasion Biology: Implications for Risk Assessment and Management of Non-Native Species. *Ethics, Policy & Environment, 19*(3), 273–284. https://doi.org/10.1080/21550085.2016.1226234

Verbrugge, L. N. H., Van Den Born, R. J. G., & Lenders, H. J. R. (2013). Exploring public perception of non-native species from a visions of nature perspective. *Environmental Management, 52*(6), 1562–1573. https://doi.org/10.1007/s00267-013-0170-1

Viana, I. G., Siriwardane-de Zoysa, R., Willette, D. A., & Gillis, L. G. (2019). Exploring how non-native seagrass species could provide essential ecosystems services: A perspective on the highly invasive seagrass *Halophila stipulacea* in the Caribbean Sea. *Biological Invasions, 21*(5), 1461–1472. https://doi.org/10.1007/s10530-019-01924-y

Vicente, J. R., Kueffer, C., Richardson, D. M., Vaz, A. S., Cabral, J. A., Hui, C., Araújo, M. B., Kühn, I., Kull, C. A., Verburg, P. H., Marchante, E., & Honrado, J. P. (2019). Different environmental drivers of alien tree invasion affect different life-stages and operate at different spatial scales. *Forest Ecology and Management, 433*, 263–275.

Vicente, J. R., Randin, C. F., Gonçalves, J., Metzger, M. J., Lomba, Â., Honrado, J., & Guisan, A. (2011). Where will conflicts between alien and rare species occur after climate and land-use change? A test with a novel combined modelling approach. *Biological Invasions, 13*(5), 1209–1227. https://doi.org/10.1007/s10530-011-9952-7

Victorian Government. (2010). *Invasive plants and animals: Policy framework.* Dept. of Primary Industries. https://agriculture.vic.gov.au/__data/assets/pdf_file/0009/582255/Invasive-Plants-and-Animals-Policy-Framework-IPAPF.pdf

Vilà, M., Bartomeus, I., Dietzsch, A. C., Petanidou, T., Steffan-Dewenter, I., Stout, J. C., & Tscheulin, T. (2009). Invasive plant integration into native plant–pollinator networks across Europe. *Proceedings of the Royal Society B: Biological Sciences, 276*(1674), 3887–3893. https://doi.org/10.1098/rspb.2009.1076

Vilà, M., Dunn, A. M., Essl, F., GÓmez-DÍaz, E., Hulme, P. E., Jeschke, J. M., NÚÑez, M. A., Ostfeld, R. S., Pauchard, A., Ricciardi, A., & Gallardo, B. (2021). Viewing Emerging Human Infectious Epidemics through the Lens of Invasion Biology. *BioScience, 71*(7), 722–740. https://doi.org/10.1093/biosci/biab047

Vilà, M., Espinar, J. L., Hejda, M., Hulme, P. E., Jarošík, V., Maron, J. L., Pergl, J., Schaffner, U., Sun, Y., & Pyšek, P. (2011). Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities and ecosystems. *Ecology Letters, 14*(7), 702–708. https://doi.org/10.1111/j.1461-0248.2011.01628.x

Vimercati, G., Kumschick, S., Probert, A. F., Volery, L., & Bacher, S. (2020). The importance of assessing positive and beneficial impacts of alien species. *NeoBiota, 62*, 525–545. https://doi.org/10.3897/neobiota.62.52793

Visseren-Hamakers, I. J. (2020). The 18th Sustainable Development Goal. *Earth System Governance, 3*, 100047. https://doi.org/10.1016/j.esg.2020.100047

Vitousek, P. M. (1986). Biological Invasions and Ecosystem Properties: Can Species Make a Difference? In H. A. Mooney & J. A. Drake (Eds.), *Ecology of Biological Invasions of North America and Hawaii* (Vol. 58, pp. 163–176). Springer New York. https://doi.org/10.1007/978-1-4612-4988-7_10

Wallace, R. D., Bargeron, C. T., & Reaser, J. K. (2020). Enabling decisions that make a difference: Guidance for improving access to and analysis of invasive species information. *Biological Invasions, 22*(1), 37–45. https://doi.org/10.1007/s10530-019-02142-2

Walton, W. C., MacKinnon, C., Rodriguez, L. F., Proctor, C., & Ruiz, G. M. (2002). Effect of an invasive crab upon a marine fishery: Green crab, *Carcinus maenas*, predation upon a venerid clam, *Katelysia scalarina*, in Tasmania (Australia). *Journal of Experimental Marine Biology and Ecology, 272*(2), 171–189. https://doi.org/10.1016/S0022-0981(02)00127-2

Warburton, B., & Anderson, D. (2018). Ecology, Economics and Ethics: The Three Es Required for the Sustainable Management of Wild Sentient Species. In S. Sarkar & B. A. Minteer (Eds.), *A Sustainable Philosophy—The Work of Bryan Norton* (pp. 237–252). Springer International Publishing. https://doi.org/10.1007/978-3-319-92597-4_14

Weston, N., Bramley, C., Bar-Lev, J., Guyula, M., & O'Ryan, S. (2012). Arafura three: Aboriginal ranger groups protecting and managing an internationally significant swamp. *Ecological Management & Restoration, 13*(1), 84–88. https://doi.org/10.1111/j.1442-8903.2011.00626.x

White, S. (2013). Into the Void: International Law and the Protection of Animal Welfare. *Global Policy, 4*(4), 391–398. https://doi.org/10.1111/1758-5899.12076

WHO. (2014). *A brief guide to emerging infectious diseases and zoonoses.* WHO Regional Office for South-East Asia. https://apps.who.int/iris/handle/10665/204722

Wilkinson, D. M. (2004). The parable of Green Mountain: Ascension Island, ecosystem construction and ecological fitting. *Journal of Biogeography, 31*(1), 1–4. https://doi.org/10.1046/j.0305-0270.2003.01010.x

Willett, K. W., Lintott, C. J., Bamford, S. P., Masters, K. L., Simmons, B. D., Casteels, K. R., Edmondson, E. M., Fortson, L. F., Kaviraj, S., Keel, W. C., Melvin, T., Nichol, R. C., Raddick, M. J., Schawinski, K., Simpson, R. J., Skibba, R. A., Smith, A. M., & Thomas, D. (2013). Galaxy Zoo 2: Detailed morphological classifications for 304 122 galaxies from the Sloan Digital Sky Survey. *Monthly Notices of the Royal Astronomical Society, 435*(4), 2835–2860. https://doi.org/10.1093/mnras/stt1458

Williams, C. E. (1997). Potential Valuable Ecological Functions of Nonindigenous Plants. In J. O. Luken & J. W. Thieret (Eds.), *Assessment and Management of Plant Invasions* (pp. 26–34). Springer. https://doi.org/10.1007/978-1-4612-1926-2_4

Williams, F., Eschen, R., Harris, A., Djeddour, D., Pratt, C., Shaw, R. S., Varia, S., Lamontagne-Godwin, J., Thomas, S. E., & Murphy, S. T. (2010). *The Economic Cost of Invasive Non-Native Species on Great Britain.* https://www.britishecologicalsociety.org/wp-content/uploads/The_Economic_Cost_of_Invasive_Non-Native_Species_to_Great_Britain1.pdf

Williamson, M. (1996). *Biological invasions* (Vol. 334). Springer Science & Business Media. https://doi.org/10.1016/j.crvi.2010.12.008

Williamson, M., & Fitter, A. (1996). The varying success of invaders. *Ecology, 77*(6), 1661–1666. https://doi.org/10.2307/2265769

Wilson, J. R. U., Dormontt, E. E., Prentis, P. J., Lowe, A. J., & Richardson, D. M. (2009). Something in the way you move: Dispersal pathways affect invasion success. *Trends in Ecology & Evolution, 24*(3), 136–144. https://doi.org/10.1016/j.tree.2008.10.007

Wilson, J. R. U., García-Díaz, P., Cassey, P., Richardson, D. M., Pyšek, P., & Blackburn, T. M. (2016). Biological invasions and natural colonisations are different – the need for invasion science. *NeoBiota, 31*, 87–98. https://doi.org/10.3897/neobiota.31.9185

Wilson, S. D., & Pinno, B. D. (2013). Environmentally-contingent behaviour of invasive plants as drivers or passengers. *Oikos, 122*(1), 129–135. https://doi.org/10.1111/j.1600-0706.2012.20673.x

Wolfe, N. D., Dunavan, C. P., & Diamond, J. (2007). Origins of major human infectious diseases. *Nature, 447*(7142), Article 7142. https://doi.org/10.1038/nature05775

WoRMS. (2022). *World Register of Marine Species.* [dataset]. https://doi.org/10.14284/170

Yang, Q., Weigelt, P., Fristoe, T. S., Zhang, Z., Kreft, H., Stein, A., Seebens, H., Dawson, W., Essl, F., König, C., Lenzner, B., Pergl, J., Pouteau, R., Pyšek, P., Winter, M., Ebel, A. L., Fuentes, N., Giehl, E. L. H., Kartesz, J., … van Kleunen, M. (2021). The global loss of floristic uniqueness. *Nature Communications, 12*(1), 7290. https://doi.org/10.1038/s41467-021-27603-y

Zarnetske, P. L., Skelly, D. K., & Urban, M. C. (2012). Biotic Multipliers of

Climate Change. *Science, 336*(6088), 1516–1518. https://doi.org/10.1126/science.1222732

Zenni, R. D., Dickie, I. A., Wingfield, M. J., Hirsch, H., Crous, C. J., Meyerson, L. A., Burgess, T. I., Zimmermann, T. G., Klock, M. M., Siemann, E., Erfmeier, A., Aragon, R., Montti, L., & Le Roux, J. J. (2017). Evolutionary dynamics of tree invasions: Complementing the unified framework for biological invasions. *AoB Plants, 9*(1). https://doi.org/10.1093/aobpla/plw085

Zenni, R. D., & Nuñez, M. A. (2013). The elephant in the room: The role of failed invasions in understanding invasion biology. *Oikos, 122*(6), 801–815. https://doi.org/10.1111/j.1600-0706.2012.00254.x

Chapter **2**

TRENDS AND STATUS OF ALIEN AND INVASIVE ALIEN SPECIES[1]

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Chapter 2

Disclaimer on maps

The designations employed and the presentation of material on the maps used in the present report do not imply the expression of any opinion whatsoever on the part of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein.

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Chapter **2**

TRENDS AND STATUS OF ALIEN AND INVASIVE ALIEN SPECIES

EXECUTIVE SUMMARY

1 **At least 39,215 alien species and more than 37,000 established alien species have been recorded worldwide and occurrences of established alien species have been reported from all countries and all ecosystems globally *(established but incomplete)* {2.2.2}.** Among these, 5,256 species have been classified as invasive according to the database underlying this chapter *(established but incomplete)* {2.2.2}. The distribution of established alien species shows marked hotspots of high species numbers, mostly located in North America, Europe, and Australasia, but also in individual African and Asian countries *(established but incomplete)* {2.2.2}. However, low data availability, particularly in Africa and Central Asia, suggests that many more unrecorded established alien species are extant but not reported due to a lack of monitoring and data integration *(established but incomplete)* {2.1.3, 2.1.4, 2.7}. Thus, the reported numbers of alien, established alien, and invasive alien species are likely severely underestimated *(well established)* {2.1.3., 2.1.4}.

2 **The number of established alien species has risen at continuously accelerating rates for centuries, recently reaching the highest total number of established alien species and highest annual rate of new records *(established but incomplete)* {2.2.1}.** The rise in established alien species numbers has had periods of uniform increases and marked accelerations *(well established)* {2.1, 2.2.1}. Before 1800, the introduction of alien species was largely driven by European colonialism, while recently introductions for ornamental purposes or associated with international transport have become more important pathways *(well established)* {2.1, 2.1.2, 2.3.1.2, 2.3.1.6, 2.4.2.2, 2.4.5.2, Box 2.5}. Marked accelerations of alien species introductions were observed circa 1800 and post-1950, currently reaching the highest value yet; 37 per cent of documented alien species introductions over the last two centuries have occurred since 1970 *(established but incomplete)* {2.1}. In addition to total numbers, the rate of increase of newly recorded alien species, which later became established, has also continuously risen with approximately 200 new alien species now recorded annually worldwide *(established but incomplete)* {2.2.1}.

3 **In absolute values, the highest numbers of established alien species records have been reported for vascular plants, insects, fishes, fungi, and molluscs *(established but incomplete)* {2.2.2}.** The distribution of established alien species worldwide is similar across taxonomic groups, with hotspots located in North America, Europe, and Australasia *(established but incomplete)* {2.2.2}. Vascular plants and mammals are the most widespread invasive alien species *(well established)* {2.2.2}. Temporal trends of records revealed three main patterns: For vascular plants, the number of records and the rate of increase rose distinctly from the nineteenth century to the present *(well established)* {2.3.2.1}, while for invertebrates, algae, and microorganisms, numbers and rates showed a marked increase particularly after 1950, likely due to increasing trade *(established but incomplete)* {2.3.1.6; 2.3.1.8, 2.3.1.9, 2.3.2.3, 2.3.3}. Mammals represent the only taxonomic group where the rate of new annual records has consistently declined since 1950, likely as a result of stricter regulations. However, while declining, the rate is still positive resulting in additional new alien mammal records each year *(established but incomplete)* {2.3.1.1}.

4 **The total numbers of established alien species are similar in all IPBES regions except for Africa, ranging from 14,797 to 17,628 established alien species in the Americas, Europe and Central Asia, and Asia and the Pacific; total numbers are distinctly lower for Africa, which hosts a maximum of 6,484 established alien species *(established but incomplete)* {2.4.1, 2.4.2, 2.4.3, 2.4.4, 2.4.5}.** The lower number of established alien species in Africa likely results from a combination of reduced introduction effort and lower data availability; therefore, the true number of alien and invasive alien species is expected to be markedly higher in Africa than currently reported *(established but incomplete)* {2.4.1}. Likewise, rates of increase were similar among the Americas, Europe and Central Asia, and Asia and the Pacific, but lower for Africa where data are less complete *(established but incomplete)* {2.4.2, 2.4.3, 2.4.4, 2.4.5, 2.7}.

5 **The majority of established alien species have been reported from terrestrial ecoregions (75 per cent), while distinctly fewer established alien species were recorded in freshwater and marine ecosystems**

***(established but incomplete)* {2.5.1, 2.5.2, 2.5.3, 2.5.4}.** In part, this pattern reflects the natural distribution of species across ecosystems. However, aquatic habitats and marine systems in particular are less thoroughly sampled in comparison to terrestrial systems, suggesting that many more alien marine species have not been detected and recorded *(established but incomplete)* {2.5.2, 2.5.3, 2.5.4}.

6 The number of established alien species is expected to rise further with a predicted 36 per cent global increase by 2050, but with large variations by region and among groups of organisms; most existing established alien species are expected to expand their current ranges *(established but incomplete)* {2.6.1}. Annual rates of increase are predicted to rise further for invertebrates, such as insects and molluscs, likely as a consequence of anticipated increasing trade and transport, but to decline for mammals, probably due to efforts to prevent their introduction and spread *(established but incomplete)* {2.6.1}. However, models and scenarios to project biological invasion dynamics are scarce and underdeveloped, hindering a robust assessment of future dynamics *(well established)* {2.6.5}. Although some established alien species have reached their geographic range limits, most established alien species are likely to further expand their alien ranges in the near future *(established but incomplete)* {2.6.1}.

7 The number of established alien species is consistently lower on land managed by Indigenous Peoples *(established but incomplete)* {Box 2.6}. Indigenous Peoples' lands are often remote and host more natural habitats compared to other lands, but that has not protected them from alien species introductions. A total of 6,351 established alien species and 2,355 invasive alien species have been recorded worldwide on Indigenous Peoples' land *(established but incomplete)* {Box 2.6}. Hotspots of biological invasions on Indigenous lands with high numbers of established alien species are found on all inhabited continents but especially in Australasia, North America, and Europe *(established but incomplete)* {Box 2.6}, regions that have the highest established alien species numbers in general. Invasive alien species affect the livelihoods and good quality of life of Indigenous Peoples and local communities worldwide *(established but incomplete)* {Box 2.11}. However, most available studies on lands of Indigenous Peoples and local communities and on good quality of life focus on woody vascular plants, while much less information is available for the effects of other taxa, particularly microbes and insects *(established but incomplete)* {Boxes 2.6 and 2.11}.

8 Islands generally host high numbers of alien and invasive alien species *(well established)* {Box 2.5}. Compared to mainland areas, the number of established alien species on islands is often very high *(well established)* {Box 2.5}. For vascular plants, the numbers of established alien species exceed the total number of native species on many islands, doubling the plant species richness on those islands *(well established)* {Box 2.5}. Worldwide, widespread invasive alien species on islands include mammals such as *Rattus* spp. (rats), *Mus musculus* (house mouse), and *Felis catus* (cat), and plants such as *Leucaena leucocephala* (leucaena), *Lantana camara* (lantana), and *Ricinus communis* (castor bean) *(well established)* {Box 2.5}.

9 Research intensity and data availability documenting established alien species' occurrences have increased in recent decades, but information about alien species distributions remains incomplete, particularly for inconspicuous species such as invertebrates, microorganisms, and aquatic species *(well established)* {2.1.4, 2.2.2, 2.7}. Lists of established alien species occurrences are very likely incomplete in the vast majority of cases across in the world *(established but incomplete)* {2.1.3, 2.1.4}. There are, however, major critical gaps for many species groups in large parts of Africa and Central Asia, for invertebrates and microorganisms, and for marine and freshwater species worldwide *(well established)* {2.2.2, 2.3.1.11, 2.3.2.5, 2.3.3.3, 2.4.2.5, 2.4.5.5, 2.5.1}. Gaps in recording alien species occurrences result in incomplete alien species lists and prevent a fully comprehensive assessment of the trends and status of invasive alien species across all taxa and habitats *(established but incomplete)* {2.2.2}. Further uncertainty arises from time lags that can span several decades from species introductions to their first detection *(well established)* {2.2.1, 2.2.3}, very likely making the documented numbers of established alien species a severe underestimate of the true extent of biological invasions *(well established)* {2.2.1, 2.2.2}. Importantly, incomplete data does not preclude drawing robust conclusions about alien and invasive alien species *(well established)* {2.7}. By taking data uncertainty into account, experts can provide a complete, credible, and transparent assessment that can be updated as more information becomes available *(well established)* {2.7}.

10 A global assessment of biological invasions that covers the trends and status of regions and species groups equally can be achieved by a major increase in efforts to monitor alien and invasive alien species and by standardizing protocols for handling and sharing data at a global scale *(established but incomplete)* {2.7}. Closing knowledge gaps in all regions and species groups and improving understanding of biotic and abiotic interactions that influence how species respond to environmental changes can be achieved through consistent, repeatable, and comparable studies of alien species occurrences that are deposited into publicly available repositories *(established but incomplete)* {2.7}. Additional applications of technology (e.g., remotely sensed data, environmental DNA) applied at large spatial scales can also provide comprehensive coverage of alien and invasive alien species *(established but incomplete)* {2.7}. Engagement by and with policymakers, citizen scientists, and Indigenous Peoples and local communities worldwide is critical to close data and knowledge gaps *(established but incomplete)* {2.7}.

2.1 INTRODUCTION

Assessing current and future dynamics of biological invasions requires data and knowledge on the geographic extent of invasive alien species, which can be used to identify hotspots of invasive alien species (**Glossary**). Further, a more comprehensive assessment depends on information about temporal trends (**Glossary**) to evaluate past and potential future species spread and detailed information on alien species, which while not yet classified as invasive in certain regions could become invasive in the future. To achieve a comprehensive global assessment of biological invasions, this chapter includes information on temporal trends and spatial distributions of both **alien** and **invasive alien species** (a subset of alien species).

Humans have introduced species to regions outside of their native ranges (**Glossary**) for millennia, and throughout, these introductions have undergone different periods of acceleration. As early as approximately 8000 B.C., neolithic people unintentionally distributed plant seeds when transporting crops (e.g., Di Castri, 1989). The first evidence of agricultural crops being traded over long distances comes from the Pharaohs of ancient Egypt approximately 3,000 to 1,500 years ago (Janick, 2007) and from Mesoamerica around the same period (Sanchéz, 1997). While early reports are scarce and inaccessible, evidence of increasingly frequent species exchanges has accumulated. The intensity of biotic exchange is often related to the extent and power of a particular empire, such as the Romans, Greeks, Aztecs, Polynesians, or the Han Dynasty. All introduced a variety of species throughout their reigns that continue to survive in their new locations (P. A. Cox & Banack, 1991; Di Castri,

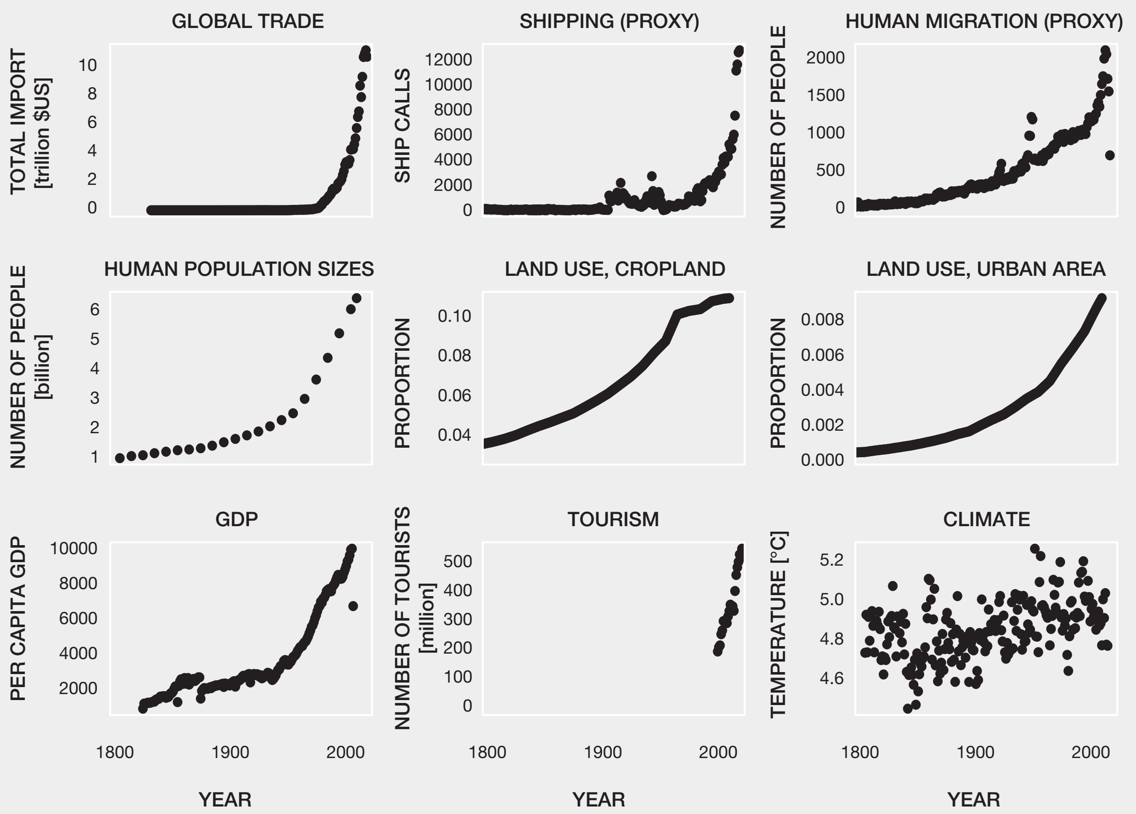

Figure 2 1 **Trends in drivers of change in nature and correlates of biological invasions.**

Panels show temporal trends of a selection of main drivers and correlates of biological invasions averaged globally. For "shipping" and "human migration" only proxy variables are shown due to the lack of more comprehensive data covering the full time period. Although these proxy variables represent only subsets of the full dynamics, they well indicate the overall temporal patterns of change. A data management report for this figure is available at https://doi.org/10.5281/zenodo.7615582

1989; Ma *et al.*, 2003; Sanchéz, 1997). As these empires expanded and the capacity of humans to travel long distances improved, there was a concomitant rise in the magnitude of alien species introductions.

The establishment of sea routes between Europe, the Americas, Africa, and Asia in the fifteenth century marked the onset of a truly global trade network that facilitated a continuously growing rise in alien species introductions (**Figure 2.1**; Di Castri, 1989) but the extent of increase varied considerably between taxonomic groups and geographic regions. Nonetheless, there has been a marked intensification of alien species exchanges across all taxonomic groups and regions in the last 200 years; the nineteenth century and post-1950s eras experienced especially high increases of new species introductions, i.e., 37 per cent of all documented established alien species introductions have occurred since 1970 (Bonnamour *et al.*, 2021; Seebens, Blackburn, *et al.*, 2017). Given the incomplete and inconsistent records of documented historic introductions, it is likely that past introduction rates were even higher (Seebens, Blackburn, *et al.*, 2017).

While many species have been unintentionally introduced, other introductions in the pre-historic, historic, and modern eras have been intentional, occurring for purposes including food, horticulture, sport hunting and fishing, the fur trade, the pet trade, and for nature's contributions to people such as erosion control and biological control (**Glossary**; e.g., Eviner *et al.*, 2012; Genovesi *et al.*, 2009; Luken & Thieret, 1997; R. M. Pringle, 2005; Reichard & White, 2001; Simberloff, 2012). The introduction pathways (**Glossary**) and the taxa introduced have varied over time **(Table 2.1; Figure 2.2)**.

The introduction of alien species is coupled with human activities and it is therefore unsurprising that invasion trends and human socio-economic activities are closely linked (Hulme, 2009; Levine & D'Antonio, 2003; X. Liu *et al.*, 2019; Meyerson & Mooney, 2007; Pyšek, Jarosik, *et al.*, 2010). Different drivers may affect invasion dynamics and become important during different stages of the biological invasion process (**Glossary**), such as the introduction and establishment stages. For instance, global trade and transport are well-known major drivers promoting the intentional or unintentional introduction of alien species (**Chapter 3**, **section 3.2.3**; and Hulme, 2009). Tourism is another important driver (**Chapter 3**, **section 3.2.3.4**), particularly on remote islands (Toral-Granda *et al.*, 2017). But interactions between introduction pathways and invasion stages also vary by taxonomic group (e.g., Bernery *et al.*, 2022). Anthropogenic disturbances such as habitat (**Glossary**) destruction (e.g., deforestation), degradation (e.g., eutrophication) and fragmentation, and climate change are strongly associated with increasing habitat vulnerability to invasions (Hierro *et al.*, 2006; Hulme, 2017; Pauchard & Alaback, 2004; J.-Z. Wan *et al.*, 2019). Thus, once introduced, alien species are more likely to establish in areas with high degrees of land use change, high human population density, and high gross domestic product (GDP) (Pyšek, Jarosik, *et al.*, 2010). All of these drivers have distinctly increased in the last decades (**Figure 2.1**; **Chapter 3**, **section 3.1.1**), paving the way for rising numbers of invasive alien species, and the establishment of alien species more generally.

2.1.1 Previous alien and invasive alien species assessments

Multiple recent regional and global scale assessments have highlighted biological invasions as having a significant influence on nature (**Glossary**), nature's contributions to people, good quality of life and on Indigenous Peoples and local communities (**Glossary**; IPBES, 2018a, 2018b, 2018c, 2019a). In general, these assessments have noted that while progress has been made in identifying pathways of alien species introductions and in invasive alien species eradication and management (**Glossary**; Secretariat of the CBD, 2020), successful prevention of biological invasions (**Glossary**) remains limited, in part due to ineffective border controls in some countries (Secretariat of the CBD, 2014). Global and regional assessment reports show that biological invasions are an increasing worldwide threat (Early *et al.*, 2016; Osipova *et al.*, 2017; WWF, 2018) exerting pressure on native biodiversity in concert with other global phenomena (IPBES, 2016; Secretariat of the CBD, 2020) resulting in consequences such as biotic homogenization and the extinction of native species (**Glossary**; Millennium Ecosystem Assessment, 2005). However, both positive and negative impacts (**Glossary**) associated with alien species have been documented (IPBES, 2016; Roué *et al.*, 2017). Nonetheless, large swathes of several regions remain understudied and report relatively little information regarding invasive alien species (IPBES, 2018b). In Europe, Central Asia, and in the Americas, biological invasions are severe due to extensive trade and transportation networks that are pathways for alien species introductions (IPBES, 2018b, 2018c) with more complete documentation in Europe and North America. In Central Asia, South America and mesoamerica, and in Africa, biological invasions tend to be less well-documented and few sources on the biogeographic details of invasive alien species trends are available across these regions (IPBES, 2018a, 2018b, 2018c). Further, invasive alien species are identified by Indigenous Peoples and local communities as one of the major drivers of change in nature as, for example, these species encroach on grazing lands and threaten agricultural systems (Forest Peoples Programme *et al.*, 2020; Roué *et al.*, 2017). Many invasive alien species do not have any cultural or economic value for Indigenous Peoples and local communities and some groups lack strategies to deal with biological invasions (Roué *et al.*, 2017).

2.1.2 Pathways of alien species introductions

Following standard frameworks (CBD, 2014; Hulme *et al.*, 2008), pathways describe the mechanisms that result in the introduction of alien species. Pathways usually focus on movements until a species reaches the border of an administrative unit, such as a country, although they are not restricted to this definition. Pathways are distinct from routes of introduction; pathways describe how and by what means a species has entered the new region; route of introduction refers to a geographic route between two locations. Pathways have been categorized into six major classes (release, escape, contaminant, stowaway, corridor, and unaided) and several sub-classes. Major classes of pathways are provided by the Convention on Biological Diversity (CBD; CBD, 2014; **Table 2.1**; **Chapter 1**, **Box 1.6**).

Alien species have been introduced through a variety of pathways that have varied in importance over time and among species groups (**Figure 2.2**; CBD, 2014; Faulkner *et al.*, 2016; Hulme *et al.*, 2008; Pyšek *et al.*, 2011). Intentional introduction pathways, such as release and escape, have played a major role for plant and vertebrate introductions, while unintentional introduction pathways, such as contaminant and stowaway, are highly relevant for introduced invertebrates, algae, and fungi (Saul *et al.*, 2017). In addition to variations among species groups, the relative importance of pathways for introducing alien species and the absolute number of alien species introduced through certain pathways has changed over time depending on the number of propagules being transported (van Kleunen *et al.*, 2018). Overall, the absolute number of established alien species has increased across nearly all pathways with particularly steep increases beginning circa 1800 and continuing until the present **(Figure 2.2)**. The main pathway recorded for most species was escape from confinement, followed by contaminant and stowaway, release in nature, and corridors. The relative importance of the escape pathway has declined slightly in recent decades, while the contaminant and stowaway pathways have increased in importance, possibly reflecting higher numbers of introductions through global trade and transport (Hulme, 2009). For detailed pathway classifications, seed contamination was the only pathway with declining absolute numbers, and particularly strong increases were observed for pet species and stowaways **(Figure 2.2)**. Overall, introductions for ornamental purposes remained highest in absolute numbers over the last 200 years. However, most (82 per cent of all available records in the pathway data set by Saul *et al.* (2017)) information on pathways is available for plants and vertebrates, while information on introduction pathways is often lacking for

Table 2 1 **Definition of major pathway classes.**

Definitions are published by the CBD (2014).

Pathway class	Definition
Release in nature	The intentional introduction of live alien organisms for the purpose of human use in the natural environment. Examples include biological control, erosion control, releases for fishing or hunting in the wild, landscape “improvement” and introductions of threatened organisms for conservation or religious purposes.
Escape from confinement	The movement of (potentially) invasive alien species from confinement (e.g., zoos, aquaria, botanic gardens, agriculture, horticulture, forestry, aquaculture and mariculture facilities, scientific research or breeding programmes, or escaped pets) into the natural environment. Through this pathway, organisms were purposefully imported or otherwise transported to confined conditions, but subsequently unintentionally escaped confinement.
Transport–Contaminant	The unintentional movement of live organisms as contaminants of a commodity that is intentionally transferred through international trade, development assistance, or emergency relief. This includes pests and diseases of food, seeds, timber, and other products of agriculture, forestry, and fisheries, as well as contaminants of other products.
Transport–Stowaway	The moving of live organisms attached to transporting vessels and associated equipment and media. The physical means of transport-stowaway include various conveyances, ballast water and sediments, biofouling of ships, boats, offshore oil and gas platforms and other water vessels, dredging, angling or fishing equipment, civil aviation, sea and air containers.
Corridor	The movement of alien organisms into a new region following the construction of transport infrastructure without which spread would not have occurred. Such trans-biogeographical corridors include international canals (connecting river catchments and seas) and transboundary tunnels linking mountain valleys or oceanic islands.
Unaided	The secondary natural dispersal of invasive alien species that have been introduced by means of any of the foregoing pathways.

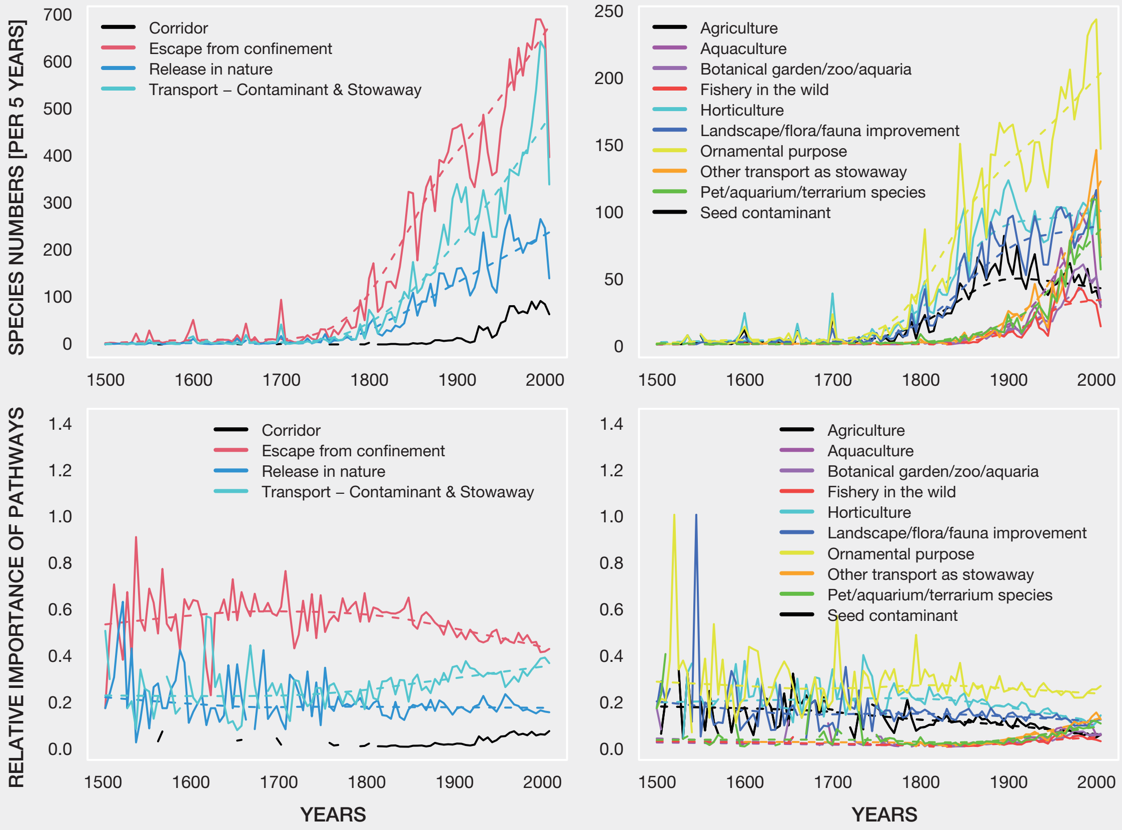

Figure 2 2 **Introductions of established alien species by pathway over time.**

The figure shows global absolute numbers (top) and relative importance (bottom) of established alien species introductions by pathway since 1500. Smoothed trends are indicated by dashed lines. Sudden drops at the end of the time series likely reflect a lack of recent records. Only the top ten pathway sub-categories are shown. A data management report for this figure is available at https://doi.org/10.5281/zenodo.7615582

other taxa. Therefore, the patterns and trends in pathway dynamics described above are likely biased towards pathways associated with plant and vertebrate introductions.

2.1.3 Chapter structure and content

Chapter 2 presents an overview of the current knowledge on the trends and status of alien species in general and invasive alien species. The logic underlying this chapter, the definitions of trends and status, and how the terms are used are presented in **Box 2.1**. Throughout the chapter, three distinct categories for species introduced to regions outside of their native ranges have been used: alien species, established alien species, and invasive alien species (**Chapter 1**, **Figure 1.1**, **Glossary**). These three status categories have been included because studies and databases vary in their definitions and details for these terms, some studies address only alien species without further specification, others focus on established alien species, while others distinguish among alien, established alien, and invasive alien species. It is critical to distinguish the status categories of species along the process of biological invasions for two main reasons, that is, because each term has a distinct meaning in invasion science and because the introduction dynamics, species distributions, and factors driving invasion patterns vary by taxa (Hejda *et al.*, 2009). The ability to clearly delimit invasive alien species from established alien species is impacted by a lack of standardized definitions systematically applied across studies and databases. Moreover, the status of a species introduced outside of its native range can change at any given time, further complicating assessments. Consequently, it remains difficult to consistently and comprehensively collate information on invasive alien species trends and status only; thus, alien and established alien species are also considered. This chapter does include one figure depicting temporal trends of invasive alien species numbers (**Figure 2.4**, in **section 2.2.1**) and multiple tables of the most widespread (**Glossary**) invasive alien species as provided by the Global Register of Introduced and Invasive Species (GRIIS; Pagad *et al.*, 2022). However, most available information and data are for established alien species. When known, the specific invasion status is therefore indicated throughout the chapter.

The structure of the chapter is depicted in **Figure 2.3**. This chapter reports on trends, status, and gaps consistently across all major sections. The major sections represent first a general introduction (**section 2.1**) and an overview of the global dynamics (**section 2.2**) followed by trends, status, and gaps by taxonomic group (**section 2.3**), IPBES regions and subregions (**section 2.4**), IPBES units of analysis (**section 2.5**), and future projections (**section 2.6**). While this structure creates some redundancies, it provides comprehensive and focused information for readers interested in a particular group, system, or region.

Box 2 1 Rationale of the chapter.

Chapter 2 reports on past and future temporal trends in alien species (including established and invasive alien species where possible) numbers, their current and future status, and data and knowledge gaps for taxonomic groups, Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) regions, and units of analysis (**Chapter 1**, **sections 1.6.4** and **1.6.5**). Temporal trends are long-term directional changes over long time periods (i.e., decades to centuries) in numbers of species, populations, or individuals introduced or in the spatial extent of colonization. Trends are presented as numbers of species (species richness) and rates of accumulation over time (i.e., numbers of newly recorded established alien species per unit time). Status is the current established alien species number and distributions in a certain area such as IPBES regions (**section 2.4**) or units of analysis (**section 2.5**) – and is indicated by established alien species number per spatial unit (global, regional, and biogeographic). Data and knowledge gaps describe missing or unavailable information or data for species or taxonomic species concepts, IPBES regions, or units of analysis.

Guiding questions:

- What is the status of alien species globally, regionally, by taxon and by unit of analysis?
- What are the trends for established alien species globally, regionally, by taxon, and by unit of analysis?
- What are the data and knowledge gaps for alien species-related data and how do they vary globally, regionally, by taxon and by unit of analysis?
- What are the eco-evolutionary dynamics of biological invasions?
- What are the methodological limitations and uncertainties in future dynamics in invasive alien species?

Keywords: alien species, established alien species, invasive alien species, distribution, status, trends, data gaps

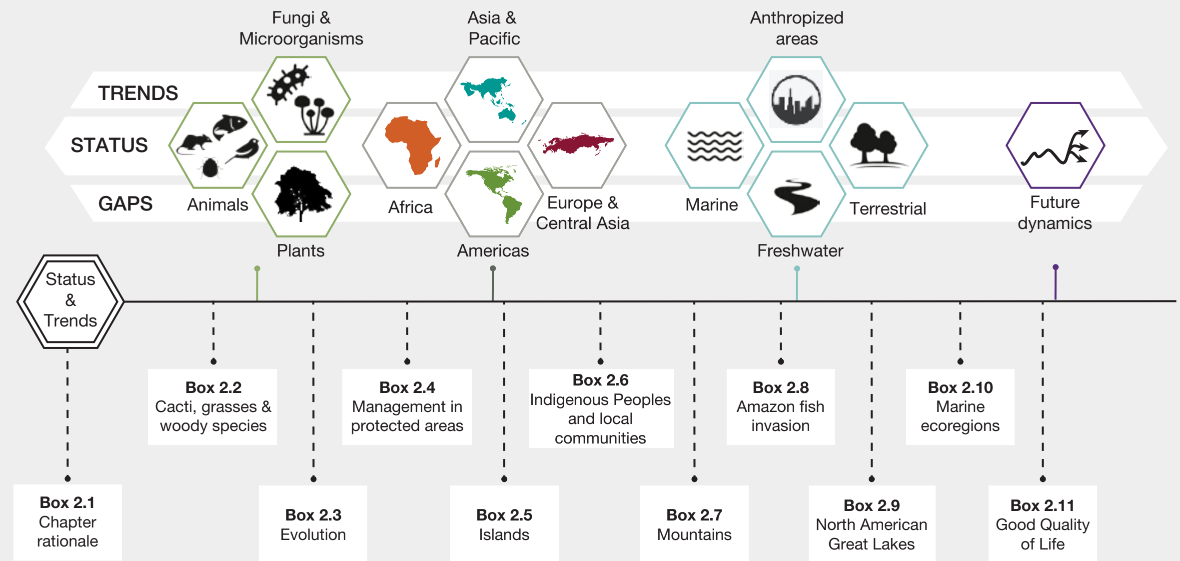

Figure 2 3 Overview of chapter structure.

Chapter 2 reports on temporal trends, the status of the current distributions of alien and invasive alien species, and the gaps in knowledge for taxonomic groups, IPBES regions, units of analysis, and future dynamics. Case studies and in-depth presentations are provided in boxes throughout the chapter.

In addition, particular emphasis was given to selected topics of overall importance in individual boxes. Throughout the chapter the term "species" is used for clarity, though it should be noted that individual populations of a species, not the entire species, are invasive. Where appropriate, the distinction has been made between major species groups, namely mammals, birds, fishes, reptiles, amphibians, insects, spiders, crustaceans, molluscs, other invertebrates, vascular plants, aquatic vascular plants, algae, bryophytes, fungi, Chromista, bacteria, and viruses.

The trends and status of alien species as presented here are based on a comprehensive review of the existing literature and databases, supplemented by knowledge from experts from all around the world and from multiple biological disciplines. The authors strove to provide a globally and taxonomically balanced and comprehensive assessment of the trends and status of alien, established alien, and invasive alien species based on available knowledge and data. However, the information residing in alien species records occurrences is scattered and patchy. A large number of records for alien species occurrences are missing for multiple reasons such as data not being publicly available, delays entering records into available databases, lack of such databases at all, or few or no monitoring activities (**Glossary**), which is particularly problematic for certain taxa such as microorganisms and sub-regions such as Central Africa. Consequently, the numbers presented in figures and tables inevitably underestimate the true numbers of alien species occurrences. However, incomplete data does not imply that inferred conclusions are flawed; instead, it means that conclusions should be drawn carefully while considering the availability and potential biases of information. In this assessment of trends and status of biological invasions, the uncertainty due to incomplete data to provide robust conclusions that are scientifically supported by currently available evidence has been included.

2.1.4 Generation of data underlying figures and tables in this chapter

Due to the use of inconsistent terminology and data processing steps, a direct comparison of individual studies of alien species occurrences is often difficult. Comprehensive global databases that allow direct comparisons of numbers across taxonomic groups and regions exist for a few well-investigated species groups. These global databases provide comprehensive information at least for individual species groups and form the basis for a database generated for this chapter.[2] All numbers presented in the tables and figures in this chapter are based on this single database compiled specifically for this chapter if not stated otherwise. Consequently, the textual descriptions of the chapter provide a more comprehensive assessment of the existing literature for the respective geographic unit or taxonomic group, while the figures and tables provide a basis for comparison across regions and taxa, which is inevitable based on a reduced number of records. The generation of the chapter database is described in detail below, and also provided in the data management report for this chapter.[2]

Generation of a database of regional checklists of alien species

The chapter database of alien species occurrences that provides the basis for figures and tables in this chapter[2] was established by integrating major global databases of alien species occurrences. These databases were selected because they are global, represent the most comprehensive databases in their field, and are published and freely accessible. Altogether, seven databases fulfilled these criteria **(Table 2.2)**: five databases with a focus on individual taxonomic groups, and two cross-taxa databases, one of which contains years of first records of alien species. The development of these databases is based on more than 4,000 individual sources of information including scientific publications, reports, and regional databases. That is, although only seven databases are included, the total number of considered publications and data sources is considerably larger. Nonetheless, it is likely that even for the species groups and content included in the databases, not all available reports and studies were considered, and records are missing for a variety of reasons. As a consequence, the numbers of species reported in figures and tables of this chapter are likely higher.

The seven global databases used as the basis for all figures and tables in this chapter differ in their spatial resolutions, terminologies, and taxonomies, impeding the direct integration of databases.[2] Assessment experts have therefore applied a workflow (i.e., a series of data transformation steps implemented in open-source computer scripts) to first standardize the spatial resolutions, terminologies, taxonomies, and the representation of years of first record. Synonyms were resolved according to the backbone taxonomy of the Global Biodiversity Information Facility (GBIF). Subsequently, the databases were combined, duplicated entries were removed, and conflicting entries, such as deviating first records, were resolved where possible. Conflicting entries that could not be resolved automatically, such as deviating invasion status, were kept as duplicated entries in the chapter database.[2] New workflows were developed to enable the identification of the biogeographical status of occurrence records using probabilistic frameworks (e.g., Arlé *et al.*, 2021).

2. The full workflow, including detailed descriptions and manuals, has been published (Seebens, 2021; Seebens *et al.*, 2020). Version 1.3.9 of the workflow (https://doi.org/10.5281/zenodo.5562840) has been applied to produce the final database version 2.4.1, which is used in this chapter (https://doi.org/10.5281/zenodo.5562892). The data management report is also available at https://doi.org/10.5281/zenodo.7615582

Table 2 2 **List of databases of alien and invasive alien species considered as a basis for figures and tables in this chapter.**

Database	Content used here	Citation and source
Global Naturalized Alien Flora (GloNAF)	Regional records of alien vascular plants	van Kleunen *et al.*, 2019 https://idata.idiv.de/DDM/Data/ShowData/257
Global Avian Invasions Atlas (GAVIA)	Regional records of alien birds	E. E. Dyer, Redding, *et al.*, 2017 https://doi.org/10.1038/sdata.2017.41
Distribution of Alien Mammals (DAMA)	Regional records of alien mammals	Biancolini *et al.*, 2021 https://doi.org/10.6084/m9.figshare.13014368
Alien amphibians and reptiles	Regional records of alien amphibians and reptiles	Capinha *et al.*, 2017 https://doi.org/10.1111/ddi.12617
MacroFungi	Regional records of alien macro fungi	Monteiro *et al.*, 2020 https://doi.org/10.15468/2qky1q
Alien Species First Records (FirstRecords)	First records of alien species in regions across taxonomic groups	Seebens, Blackburn, *et al.*, 2017 https://doi.org/10.5281/zenodo.4632335
GRIIS	Regional records of alien and invasive alien species across taxonomic groups	Pagad *et al.*, 2022 https://doi.org/10.5281/zenodo.6348164

The integration of the seven global databases as described above resulted in the largest single database of alien species distributions currently available, containing 175,980 records of 39,215 alien taxa from 264 locations worldwide. The term "location" mostly refers to countries, but the database also contains information about sub-national units such as islands or federal states in some cases. The database also includes populations with unconfirmed or "casual" (**Glossary**) status. Records of casual species are not reported in this chapter and therefore excluding casual alien species resulted in 37,591 established alien species and 5,260 invasive alien species as classified by the database GRIIS.

The databases underlying the chapter database differ in their terminology describing biological invasion status (i.e., introduced, established, invasive) of a population (Groom *et al.*, 2019). However, invasion status is often difficult to determine due to the lack of protocols for a standardized determination. Some databases, such as GloNAF, have a more rigorous and conservative approach to classifying established alien species, while other databases such as GRIIS included more species in this category. Consequently, the total numbers of established alien species vary among databases. Comprehensive global databases exist for mammals, birds, and vascular plants. These underwent a thorough assessment of invasion status and thus usually report lower numbers relative to cross-taxonomic databases such as the GRIIS or FirstRecords. To account for this variation in this assessment, total numbers of established alien species were provided as ranges for these taxonomic groups to emphasize the variation that exists in the published material. However, the spatial variations of the taxonomic databases are highly correlated with the variation in the GRIIS: The Pearson correlation coefficients, r, of total established alien species per region between GRIIS and GloNAF (r=0.92), Global Avian Invasions Atlas (GAVIA) (r=0.76) and Distribution of Alien Mammals (DAMA) database (r=0.82) were all high and significant. Thus, the spatial and temporal patterns as shown in this chapter do not distinctly differ among databases except in the overall levels of species numbers. This chapter therefore shows the total numbers of established alien species, including all databases in maps and time series, and provides ranges in tables of established alien species numbers.

Generation of a database of local occurrence records

The database used in this chapter provides information on alien species occurrences in so-called checklists representing lists of species for countries, large islands or other sub-national regions. This is inconvenient when it comes to the analysis of the distribution of alien species at other delineations such as units of analysis or marine ecoregions. To obtain information about alien species occurrences at different levels of spatial organization and scale, a freely available workflow to downscale regional checklists of alien species occurrences was applied (Seebens & Kaplan, 2022b). Using this workflow, coordinates of species occurrences as reported in the chapter database were obtained from GBIF and the Ocean Biodiversity Information System (OBIS). For each species in the chapter database, coordinates of records (marine or terrestrial) were obtained from the aforementioned online platforms and identified as representing alien populations based on the chapter database. Various steps of data cleaning and testing were included to avoid false entries. In this way, more than 35 million records of alien populations of 17,424 established alien species with coordinate-based records were gathered. These point-wise occurrence records were then aggregated to obtain total established alien species numbers per terrestrial region, marine ecoregion (see next paragraph for details, see also **Chapter 1**, **section 1.6.4** for a description of IPBES regions and sub-regions used in the IPBES invasive alien species assessment), and land managed by Indigenous Peoples (**Box 2.6** in **section 2.4.1**). The full database of coordinates is open access (Seebens & Kaplan, 2022a), and includes a manual for data generation and digital object identifiers for GBIF requests to ensure reproducibility and transparency.

Marine records

Comprehensive information about the global occurrence of marine alien species was largely lacking when work on this chapter was initiated. Since then, two important developments have taken place, namely the publication of a worldwide study on marine alien species distributions (Bailey *et al.*, 2020) and the publication of the World Register of Introduced Marine Species (WRiMS; M. J. Costello *et al.*, 2021). In both cases, records of marine alien species have been validated by experts in the field. A total number of 1,442 marine alien species were recorded by Bailey *et al.* (2020), while 2,714 species were reported by M. J. Costello *et al.* (2021). Both are likely underestimates of the true extent of marine alien species. Due to the lack of more detailed data and/or available expertise to check individual records and regions, the studies cover either only approximately half of the world's marine ecoregions or provide information on comparatively large spatial units rendering a comparison of marine ecoregions difficult. To provide an alternative way of gathering information, this assessment used the database of local occurrence records of established alien species as described in the previous paragraph, which is based on regional checklists of established alien species and records from GBIF and Ocean Biodiversity Information System (OBIS) as described in the published workflow (Seebens & Kaplan, 2022b). The coordinate-based records were then assigned to the marine ecoregion as presented by Spalding *et al.* (2007). The spatial representation is still biased towards well-investigated regions and records are not cross-checked by experts, but the generated data do provide an overview across nearly all marine ecoregions worldwide. To consider the published data validated by experts, the information provided in Bailey *et al.* (2020) has been used where possible and filled in missing regional information by the aforementioned data generation methods.

Quantification of data gaps

The lack of information on alien and invasive alien species occurrences means that regional lists (i.e., checklists) of established alien species are often incomplete, producing data gaps. The degree of incompleteness varies by taxonomic group, region, and time period (Pyšek *et al.*, 2008). To assess the influence of data gaps on the trends and status presented in this chapter, this assessment attempted to quantify the degree of incompleteness. As little research has been done previously to assess incompleteness, three different indicators of data gaps were tested:

1. The number of studies available per region in the chapter database was used as a proxy measure for research intensity and should negatively relate to data gaps.

2. To measure data gaps across taxonomic groups, the number of widespread phyla for which no information was available for a particular region was counted. A widespread phylum is defined as one with more than 500 records in the chapter database. Seven phyla were determined to be widespread: Ascomycota, Annelida, Basidiomycota, Mollusca, Chordata, Arthropoda, and Tracheophyta. Different cut-off values (other than 500 records) for selecting taxonomic groups were tested but did not change the overall patterns. The number of these phyla with less than five records per region was then counted. By applying this approach, experts assumed that at least five established alien species per selected phylum (i.e., at least five species of Tracheophyta per region, five established alien species of Arthropoda, etc.) should be found in each region as defined in the chapter database. This is likely true, particularly for large regions, but might be critical for very small regions and small islands. Different versions of this indicator were tested using different cut-off values (e.g., at least one, three, or ten records) but all versions revealed similar spatial patterns of research intensity and data gaps (**Figure 2.5** for a spatial representation of indicators 1 and 2).

3. A third indicator was used to describe spatial variation of data gaps for individual taxonomic groups by comparing the number of available first records of established alien species for a region with the total number of species recorded for the same region. This analysis provided information on the proportion of available first records per region and can be used to assess the robustness of temporal trends and provide indications about the general availability of information for the respective taxonomic group. As the biases known for first records largely reflect data and knowledge gaps in general, the proportion of available temporal information is used as a proxy for data completeness.

Although none of these indicators are ideal, they can be considered for context when interpretating the trends and status of biological invasions.

2.2 GLOBAL TRENDS AND STATUS OF ALIEN AND INVASIVE ALIEN SPECIES

This section describes an assessment of the temporal trends and status of the distribution of alien and invasive alien species globally for all taxonomic groups combined.

2.2.1 Trends

Overall, studies on the introduction of alien species over time have reported a continuous global increase in the number of established alien species consistent across taxonomic groups, particularly since the early nineteenth century (Aukema *et al.*, 2010; C. Chen *et al.*, 2017; E. E. Dyer, Cassey, *et al.*, 2017; S. Henderson *et al.*, 2006; Peck *et al.*, 1998; Pyšek *et al.*, 2012; Roy, Preston, *et al.*, 2014; Sandvik, Dolmen, *et al.*, 2019; Sax & Gaines, 2008; Verloove, 2006; Wilson *et al.*, 2007). Indeed, there is no study reporting a decline in established alien species numbers except for a few islands where eradication programmes or stringent biosecurity (**Glossary**) measures have been applied (Simberloff *et al.*, 2013). Distinct increases in established alien species numbers are often reported post-1950 (Huang *et al.*, 2011; Peck *et al.*, 1998; Pyšek *et al.*, 2012; Sandvik, Hilmo, *et al.*, 2019), while a few other reports indicate earlier acceleration in the nineteenth century (mostly for vascular plants; C. Chen *et al.*, 2017; S. Henderson *et al.*, 2006; Seebens, Blackburn, *et al.*, 2017; Wilson *et al.*, 2007) or continuous increases without periods of acceleration over 200 years (mostly for insects; Aukema *et al.*, 2010; Nahrung & Carnegie, 2020) and birds (Blackburn *et al.*, 2015). In addition to the rise in cumulative established alien species numbers, many studies also report rising rates of increase over time (Blackburn *et al.*, 2015; Seebens, Blackburn, *et al.*, 2017). Recently, the highest global emergence rates of new established alien species were reported with approximately 200 new alien species, which later became established, recorded annually (Seebens, Blackburn, *et al.*, 2017). Declining rates of new records of terrestrial alien species were observed only for vascular plants in North America (Seebens, Blackburn, *et al.*, 2017), insects in Australia (Nahrung & Carnegie, 2020) and mammals worldwide (Seebens, Blackburn, *et al.*, 2017). As shown in the GRIIS database, numbers of invasive alien species show very similar trends over time, but with lower numbers in comparison to established alien species (**Figure 2.4**; Seebens, 2021).

Most studies on selected taxonomic groups, specific regions, or global analyses show systematic and constant increases in established alien animal species across taxonomic groups (e.g., Aukema *et al.*, 2010; Bailey *et al.*, 2020; E. E. Dyer, Redding, *et al.*, 2017; Fuentes *et al.*, 2020; Seebens, Blackburn, *et al.*, 2017). For example, bird and mammal introductions mostly occurred in three distinct phases: first, historically with the discovery and colonization of new lands by Europeans from about 1500 to 1700; second, mainly through acclimatization societies (i.e., associations that encouraged the introduction of alien species), particularly *via* European colonialism from 1700 to 1900 (e.g., Pipek *et al.*, 2015); and since the 1950s, mostly *via* global trade (Biancolini *et al.*, 2021; Cassey *et al.*, 2015; E. E. Dyer, Redding, *et al.*, 2017; Hulme, 2021; Turbelin *et al.*, 2017). In contrast to alien homoeotherms, the pet trade is the primary cause of herpetofaunal introductions, a recently spreading group (Capinha *et al.*, 2017). For insects, there are two distinct waves of accelerated introduction rates, one between 1820-1914 and one from 1969 to present, likely due to intensifying global trade and transport (Bonnamour *et al.*, 2021; Roques *et al.*, 2016). Horticulture in general including the trade for ornamental purposes represents an important pathway for the introduction of vascular plants and their pathogens (**Figure 2.2**; Hulme, 2011; van Kleunen *et al.*, 2018). In addition to the total number of introduced alien species, the rate of species accumulation also continuously increased for most taxonomic groups in recent decades (see below), indicating a long-lasting intensification of introductions. Mammals represent the only exception, showing declines in species accumulation rates since about 1950, likely a consequence of stricter regulations on animal trade and husbandry and limited source pools (Seebens *et al.*, 2018; Simberloff *et al.*, 2013).

Once established in a new location, alien species are likely to spread to new areas within the introduced range either by natural dispersal or by means of human-mediated transportation. Approximately 90 per cent of all species introduced before 1700 are found today in more than one region, indicating further spread or multiple introduction events (Seebens, Blackburn, *et al.*, 2021). Spread of an alien species usually lasts for decades to centuries (Gassó *et al.*, 2010; Roques *et al.*, 2016). Rates of inter-regional spread were already high in the nineteenth century for many taxonomic groups, and peaked at that time for vascular plants, but increased further for other taxa, particularly for birds and invertebrates (Seebens, Blackburn, *et al.*, 2021). While spread appears to be slowing for a few already widespread alien species, it is likely that the vast majority of established alien species found currently in only a few sites (Pyšek, Pergl, *et al.*, 2017; Seebens, Blackburn, *et al.*, 2021) will spread also without human assistance in the near future.

The increase in numbers of established alien species is consistent among IPBES regions **(Figure 2.4)**. Before 1800, numbers of established alien species rose more rapidly in Europe and Central Asia, although Europe by far has the most records of first year of observations. The differences in early records between Europe and Central Asia and

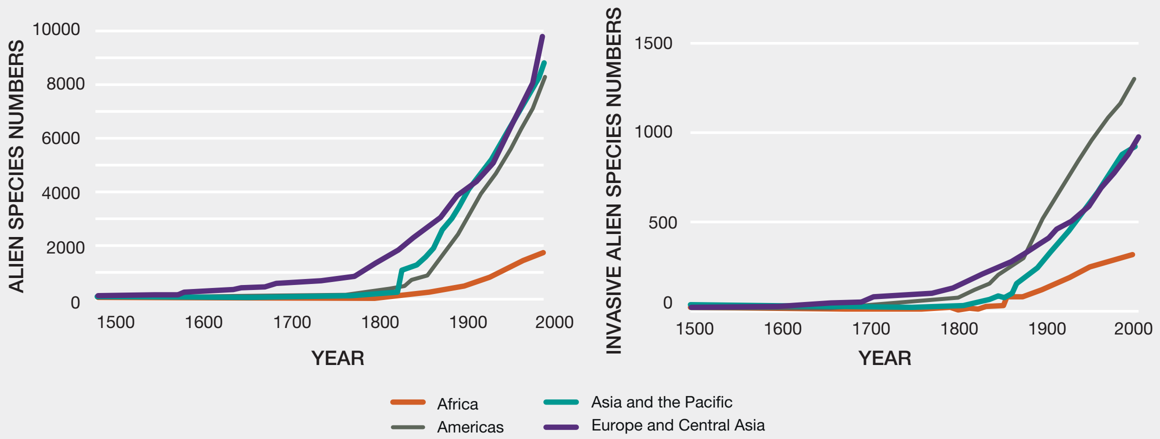

Figure 2 4 **Trends in numbers of established alien species and invasive alien species.**

Total numbers of established alien species (left) and invasive alien species (right) are shown for IPBES regions for 1500-2005. Numbers underestimate the true extent of alien species occurrences due to a lack of data (**section 2.1.4** for further details about data processing). A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

other IPBES regions are likely due to different sampling intensities (Seebens, Blackburn, *et al.*, 2017). In addition, due to time lags (lag phase in the **Glossary**), the rapid increase in researchers studying biological invasions and their impacts, and the subtlety of some impacts, the number of established alien species, and invasive alien species is almost certainly underestimated (Bellard & Jeschke, 2016). The steepest increases in established alien species were observed from post-1850 to the present, particularly for the Americas and the Asia-Pacific regions. These two IPBES regions followed similar trajectories of increases from about 1950 onwards resulting in similar total species numbers in 2005, between 7,000 and 8,000 established alien species for the Americas and the Asia-Pacific regions respectively. Note that the total number of recorded established alien species is higher than shown in the time series due to missing years of first records for most taxa and regions. The number of established alien species for Africa is notably low and markedly different from other regions. This is a general pattern that also holds when species numbers in particular taxonomic groups in Africa are plotted separately (Pyšek, Hulme, *et al.*, 2020). It is not fully understood why numbers are so much lower in Africa, but it is most likely due to Africa having lower imports than other regions, a lack of information on the year of first records of established alien species in Africa, and because the continent is generally understudied in terms of biological invasions (Pyšek *et al.*, 2008; **section 2.4.2**). As classified by GRIIS, numbers of invasive alien species show very similar dynamics though at a lower number, with correlation coefficients of times series over 0.95 for all IPBES regions **(Figure 2.4)**. The high correlation between the distribution of established alien species and invasive alien species, which has also been reported in other studies (Pyšek, Pergl, *et al.*, 2017), makes it very likely that trends and status of invasive alien species resemble those of established alien species, noting there are less invasive than established alien species.

2.2.2 Status

According to the chapter database underlying the figures and tables in this chapter, at least 39,215 alien species have been recorded worldwide. As the database does not contain all records of alien species (**section 2.1.4**), the true number is likely much higher. Of those alien species, 37,215 are recorded as having established alien populations, while 5,256 are classified as invasive alien species (**section 2.1.4**). Note that the total number of invasive alien species deviates from the number provided in **Chapter 4** due to different approaches and data sources. As the number of alien species recorded is unequally distributed across the globe **(Figure 2.5)**, because the detectable patterns depend upon available data, and because large data gaps remain (**section 2.2.3**), it is in some cases difficult to distinguish data biases and artifacts from true biological patterns. However, with continued research effort, the gaps are gradually shrinking. In the terrestrial and marine realms and consistent across taxonomic groups, the highest numbers of established alien species are found in Europe (particularly western Europe), North America, and Australasia (Dawson *et al.*, 2017). However, total numbers are higher than shown in **Figure 2.4** where only available global databases were included. For many regions, particularly several countries in Africa, Central Asia and many islands, data are scarce and available lists are incomplete. For many marine ecoregions (white areas), alien species occurrence data are lacking or not yet integrated into larger databases **(Figure 2.5)**.

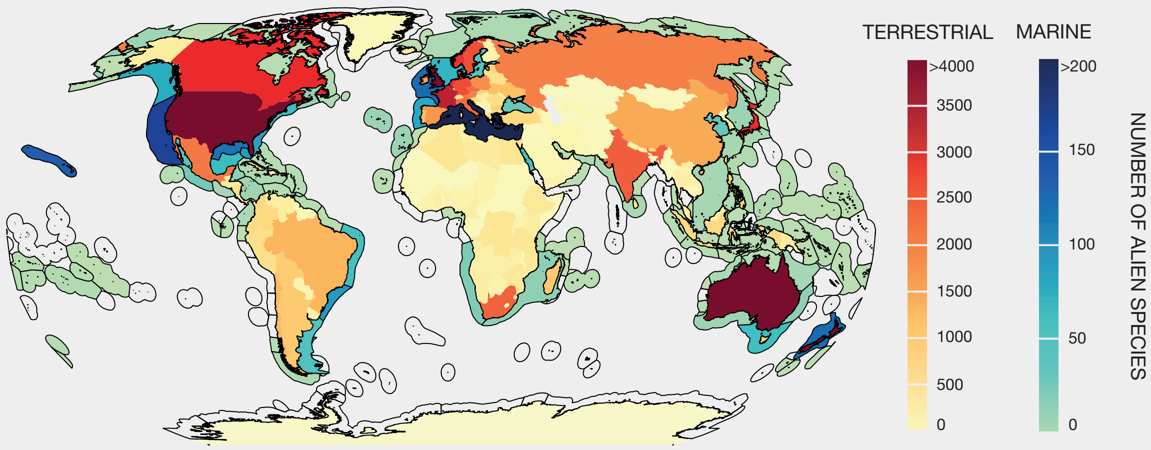

Figure 2 5 **Numbers of established alien species per region.**

The total number of established alien taxa per mainland region (terrestrial and freshwater) and marine ecoregion (marine) is indicated by colour separately. White denotes missing information. Note that marine records were available on different geographic delineations and thus marine ecoregions differ in sizes in this figure. Note that numbers may deviate from those reported in the text due to variation among data sources. See **section 2.1.4** for further details about data sources and data processing. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Global patterns of established alien species distributions were consistently assessed only for selected groups such as ants, spiders, amphibians, reptiles, freshwater fishes, birds, mammals and vascular plants for 186 islands and 423 mainland regions by Dawson *et al.* (2017). This study showed that established alien species from these groups are unevenly distributed, with some regions (particularly Europe, North America, and Australasia) harbouring more species than other regions. Although Dawson *et al.* (2017) previously provided the most comprehensive representation of established alien species distributions across taxonomic groups, their assessment included only two invertebrate groups (ants and spiders) and no marine species were included because of the lack of comprehensive information. The analysis by Dawson *et al.* (2017) based on the seven animal groups revealed two major commonalities: islands and coastal areas have greater proportions of established alien species in regional faunas, and high numbers of established alien species are associated with indicators of human activities such as land-use intensity and trade. The distribution of established alien species varies by taxonomic group. For example, biological invasion hotspots of ants are found in South America, equatorial Africa, and Southeast Asia (Bertelsmeier *et al.*, 2015), while bird and mammal invasions are concentrated in North America, western Europe, South Africa, Japan, Australia, and New Zealand (Biancolini *et al.*, 2021; E. E. Dyer, Cassey, *et al.*, 2017). Numbers of established alien species show latitudinal trends: alien bird species are greatest at mid-latitudes and reflect concomitant variations in human activity, most notably the number of species introduced to a particular location (E. E. Dyer, Redding, *et al.*, 2017). Below, overviews and examples of established alien species are provided for different taxonomic groups **(Tables 2.2, 2.3)**.

The worldwide distribution of established alien species shows a marked latitudinal gradient with the highest species numbers reported at mid-latitudes, such as the temperate regions of the Northern and Southern Hemispheres, with lower numbers in the tropics (Q. Guo *et al.*, 2021; Sax, 2001). The mechanisms that drive this pattern are not yet fully understood but may be positively correlated with invasive alien plant density, the human development index, and the location of most of well-developed countries in temperate regions (Weber & Li, 2008). Greater resistance to biological invasions, faster recovery after disturbance due to higher diversity, lack of life history traits that confer shade tolerance and lower colonization, high predation pressure, and propagule pressures (**Glossary**) are proposed, but not proven, to be major causes of lower alien richness in tropical continental regions compared to non-tropical regions (Fine, 2002; Freestone *et al.*, 2011; Isbell *et al.*, 2015; Rejmanek & Richardson, 1996). However, on islands the pattern is very different, with tropical islands harbouring very high numbers established alien species (Moser *et al.*, 2018; Rejmanek & Richardson, 1996). Thus, it seems unlikely that tropical regions have a greater resistance to biological invasions compared to non-tropical regions as they lack the characteristics to make them less vulnerable (Chong *et al.*, 2021). However, one explanation for lower numbers of established alien species in tropical regions is lower levels of propagule pressure (i.e., fewer introductions and/or smaller introduction size) due to factors such as low import volumes. In addition, reduced sampling intensities due to

Table 2 3 **Numbers of established alien species for various taxonomic groups worldwide.**

Species numbers can vary depending on data sources. Note numbers in this table may deviate from those reported in the text due to variation among data sources. For mammals, birds, and vascular plants, ranges of values indicate variation among databases (**section 2.1.4** for further details about data sources and data processing). A data management report for the data underlying this table is available at https://doi.org/10.5281/zenodo.7615582

Taxonomic group	Number of species
Mammals	197-368
Birds	495-877
Fishes	1,451
Reptiles	411
Amphibians	135
Insects	6,795
Arachnids	500
Molluscs	826
Crustaceans	661
Vascular plants	13,081-18,543
Algae	734
Bryophytes	88
Fungi	1,149
Oomycetes	70
Bacteria and protozoans	38

Table 2 4 **Top 10 most widespread invasive alien species worldwide.**

The number of regions where a species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien species and not their impacts, covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Organism group	Taxon	Number of regions
Vascular plant	*Pontederia crassipes* (water hyacinth)	74
Vascular plant	*Lantana camara* (lantana)	69
Mammal	*Rattus rattus* (black rat)	60
Vascular plant	*Leucaena leucocephala* (leucaena)	55
Mammal	*Mus musculus* (house mouse)	49
Mammal	*Rattus norvegicus* (brown rat)	48
Vascular plant	*Ricinus communis* (castor bean)	47
Vascular plant	*Ailanthus altissima* (tree-of-heaven)	46
Vascular plant	*Robinia pseudoacacia* (black locust)	45
Vascular plant	*Chromolaena odorata* (Siam weed)	43

lower research efforts and fewer monitoring programmes also likely contribute to the lower numbers recorded in the tropics (Chong *et al.*, 2021).

Comprehensive overviews of the global distribution of individual taxonomic groups exist mostly for vascular plants (E. J. Jones *et al.*, 2019; Pyšek, Pergl, *et al.*, 2017) and vertebrates (mammals, birds, amphibians, reptiles and fishes) (Capinha *et al.*, 2017; Dawson *et al.*, 2017; E. E. Dyer, Cassey, *et al.*, 2017; Pyšek, Hulme, *et al.*, 2020), with the exception of a few invertebrate groups such as spiders and ants (Dawson *et al.*, 2017) and land snails (Capinha *et al.*, 2015), and bryophytes (Essl *et al.*, 2013). Patterns of spatial distribution were similar across most taxonomic groups with particularly large numbers of terrestrial alien species in Europe, North America, and Australasia (Dawson *et al.*, 2017). As an exception, there are large numbers of alien fern species in the tropical regions of South America and Asia (E. J. Jones *et al.*, 2019). Common explanations for the variations observed in the spatial distribution of terrestrial alien species include variation in drivers such as trade and transport, GDP, high human population densities, and the degree of disturbance (Capinha *et al.*, 2017; Dawson *et al.*, 2017; E. E. Dyer, Cassey, *et al.*, 2017). Often alien species originate from neighbouring regions or regions connected through trade over long distances (D. S. Chapman *et al.*, 2017; L. Henderson, 2006; Pyšek *et al.*, 2012). High numbers of terrestrial alien species were often found on islands compared to mainlands, with remote islands often showing particularly large alien species numbers (Blackburn *et al.*, 2008; Moser *et al.*, 2018). While it is unknown whether these high numbers can be explained by high propagule and colonization pressures (**Glossary**) due to human activities, or instead are a result of the traits of the native communities, both factors likely interact to affect the outcome of invasions on islands.

2.2.3 Data and knowledge gaps

Perceptions of the distribution of alien species are highly influenced by an unequal global sampling of information on alien species occurrences. For example, hotspots (**Glossary**) of alien species occurrences (i.e., areas of high alien species richness relative to other regions with similar biogeographic characteristics; Dawson *et al.*, 2017) are well-known to coincide with global hotspots of data availability and study sites (L. J. Martin *et al.*, 2012; C. Meyer *et al.*, 2015), shaping knowledge of species distributions (A. C. Hughes *et al.*, 2021). This conclusion is confirmed by the information provided in this chapter: mapping of the number of available studies, which were used to generate the underlying database of this chapter (**section 2.1.4** for further details on the data generation), revealed that regions with high level of information on alien species occurrences **(Figure 2.6)** match the hotspots of established alien species occurrences **(Figure 2.5)**. Hence, knowledge of invasive alien species occurrences is biased towards well-sampled regions such as Europe and North America and taxonomic groups such as vertebrates and plants with the majority of studies conducted in recent decades (Bellard & Jeschke, 2016; Jeschke *et al.*, 2012; Pyšek *et al.*, 2008). It remains unclear how much of the distributions of alien species and documented hotspots is affected by spatial variation in research intensity. The investigation of data availability as described in **section 2.1.4** showed extensive data gaps, particularly in large parts of Africa, Central Asia and on islands worldwide **(Figure 2.6)**.

In addition to regional biases, research intensities vary across taxonomic groups. There is considerably more information available on the distribution of alien and invasive alien species for vertebrates, particularly mammals (**section 2.3.1.1**), birds (**section 2.3.1.2**), and vascular plants (**section 2.3.2.1**) than for other taxa. In general, there are large data and knowledge gaps for invertebrates and microorganisms. While most information about invertebrates is available for insects, crustaceans, and molluscs, these data are still incomplete for many regions of the world (**sections 2.3.1.6**, **2.3.1.8**, **2.3.1.9**). Information for other invertebrate groups is extremely scarce. Globally little information is available for alien microorganisms and recorded distributions are often biased towards individual studies. Across realms, the greatest amount of information is available for terrestrial habitats (**section 2.5.1**), while information for aquatic (marine, freshwater and brackish) alien species is often lacking (**sections 2.5.2**, **2.5.3**). Consequently, the lists of alien species for individual regions are, in most cases, incomplete, even for well-sampled regions due to the lack of information about microorganisms and invertebrates, for example, and the degree of incompleteness varies highly among regions globally.

Most of the information about alien species occurrences is available at the national scale for whole countries, while information on sub-national units such as federal states, provinces, protected areas, or private land is usually lacking. Information about occurrences is particularly scarce for lands and waters managed by Indigenous Peoples and local communities **(Box 2.6)**. Furthermore, information about abundances and changes in abundances of alien populations is available only in a few cases and is not consistently recorded across regions and taxa. Additional uncertainty in the records of alien and invasive alien species occurrences arises from time delays frequently observed between the actual species introduction and its first record as a new population outside its native range (Crooks, 2005). For vascular plants, these time lags have been estimated to be on average 20 years (Seebens *et al.*, 2015), while for individual cases time delays of up to 150 years have been recorded (Kowarik, 1995b).

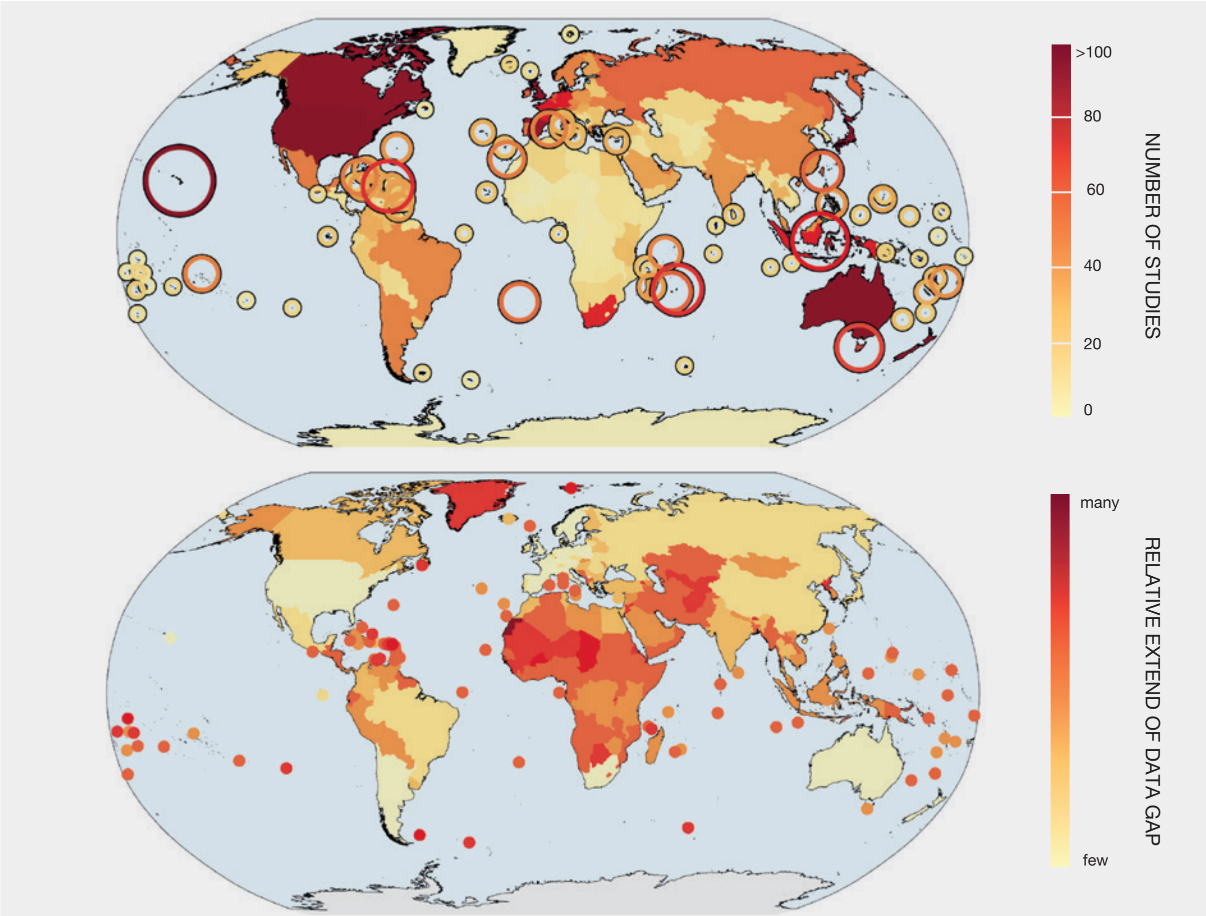

Figure 2 6 **Research intensity and data gaps for global established alien species distribution records.**

Research intensity (top) is indicated by the number of studies available in the chapter database. Data gaps (bottom) were determined as the lack of information for the seven most common phyla as recorded in the chapter database per region. Largest data gaps are apparent in Africa, Central Asia, and for many islands (**section 2.1.4** for further details about data sources and data processing for further details of the analysis). Islands are indicated by dots and circles, respectively. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

2.3 GLOBAL TRENDS AND STATUS OF ALIEN AND INVASIVE ALIEN SPECIES BY TAXONOMIC GROUPS

2.3.1 Animals

This section reports on the temporal trends and status of the distribution of alien and invasive alien animal species for various animal groups, namely mammals (**section 2.3.1.1**), birds (**section 2.3.1.2**), fishes (**section 2.3.1.3**), reptiles (**section 2.3.1.4**), amphibians (**section 2.3.1.5**), insects (**section 2.3.1.6**), arachnids (**section 2.3.1.7**), molluscs (**section 2.3.1.8**), crustaceans (**section 2.3.1.9**), and other invertebrates (**section 2.3.1.10**), as well as data and knowledge gaps (**section 2.3.1.11**).

2.3.1.1 Mammals

Trends

Because they were useful, mammals were among the first species introduced by humans, and the first records of introduced alien mammals date back thousands of years (Genovesi *et al.*, 2012). For example, mammals have been used as pack animals, for meat and fur, ornamentals, biocontrol agents, and pets since the expansion of humans from Africa to other continents (Clout & Russell, 2008; Long, 2003; Simberloff & Rejmanek, 2011). During prehistoric and historic human migration, humans transported mammals to new areas to create wild populations for settlers to hunt (Clout & Russell, 2008; Long, 2003; Simberloff & Rejmanek, 2011), peaking with European colonization. As a consequence, there were high numbers of alien mammals as early as 500-200 years ago (**Figure 2.7**). During the

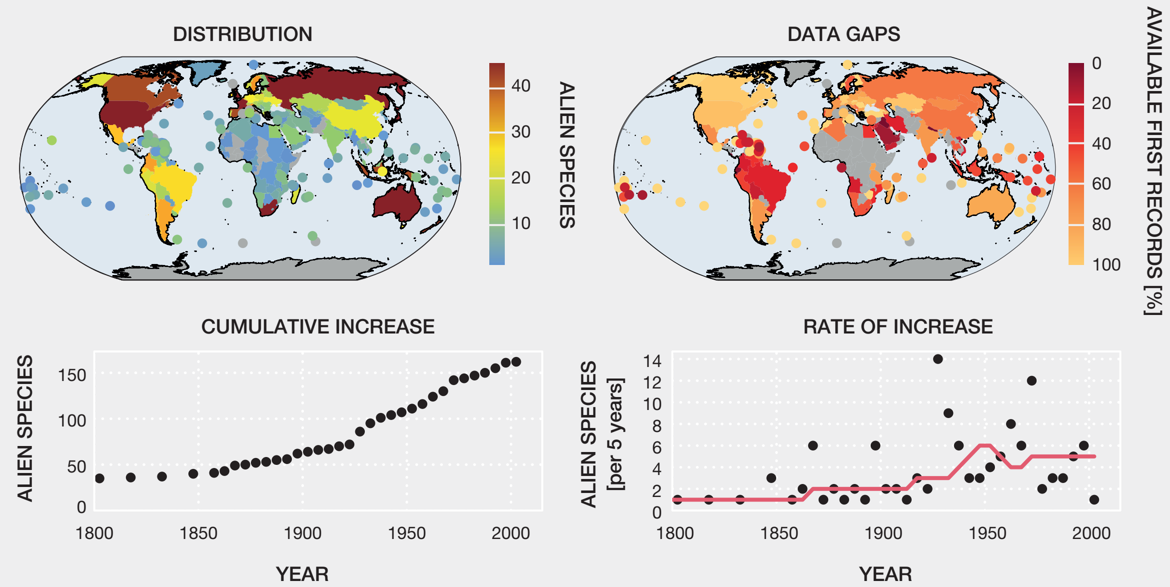

Figure 2 7 **Status, trends, and data gaps for established alien mammals.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted to aid visualization and do not indicate species numbers. Trends are shown in lower panels as cumulative numbers and as a rate of increase (i.e., numbers of alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

nineteenth century, a further acceleration of new records occurred (Biancolini *et al.*, 2021) when specific organizations (i.e., acclimatization societies) focused on alien species release to aesthetically "improve" the landscape and local fauna of colonial territories (Osborne, 2000; Simberloff & Rejmanek, 2011). In recent decades, the dominant pathways of mammal introductions have shifted from hunting and "faunal improvement" to the pet trade likely due to stricter regulations targeting alien mammals (Simberloff *et al.*, 2013). Many mammal introductions outside of their native ranges were also carried out for conservation, and to protect mammal species from overhunting, habitat loss, and invasive alien predators (Biancolini *et al.*, 2021; Seddon *et al.*, 2015; Woinarski *et al.*, 2015).

Status

The biological invasion history and status of mammals are among the best documented of alien animal taxa (Biancolini *et al.*, 2021; Blackburn *et al.*, 2017; Clout & Russell, 2008; Long, 2003). At present, 241 mammal species have established alien populations globally, causing many and diverse environmental impacts, especially on insular ecosystems (**Glossary**; Biancolini *et al.*, 2021; Blackburn *et al.*, 2017; Clout & Russell, 2008; **Chapter 4**, **section 4.3.1.1**). If the few records of unsuccessful and unconfirmed introductions are included, at least 274 mammal species have been introduced by humans to new locations (Blackburn *et al.*, 2017; Zenni & Nuñez, 2013).

According to the global Distribution of Alien Mammals database (DAMA), Asia has the highest number of established alien mammals (95), followed by North America (79), Europe (76), Australia (54), Africa (52), Oceania (50), and South America (42) (Biancolini *et al.*, 2021). The major global donors of alien mammal species are Asia (91 established alien species) and Europe (34), Australia (32), North America (31), Africa (30), and South America (23 alien species). An outgoing species flow directed to other continents is predominant for Europe and Asia, while an intracontinental flow (i.e., alien species introduced to other parts of their native continent) is common for Australia (74 per cent of all alien Australian mammals), North America (61 per cent), South America (5 per cent), and Africa (56 per cent). Other countries of Oceania received species only from other continents (Biancolini *et al.*, 2021).

Globally, the vast majority (81 per cent) of alien mammal records are found on islands (Biancolini *et al.*, 2021), most likely due to the higher vulnerability to biological invasions of insular ecosystems and greater propagule and colonization pressure on islands relative to mainland systems (Dawson *et al.*, 2017; Moser *et al.*, 2018). Moreover, alien mammals occur on 97 per cent of islands that harbour highly

Table 2 5 **Top 10 most widespread invasive alien mammal species worldwide.**

The number of regions where a species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien mammal species, not impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Taxon	No. of regions	Taxon	No. of regions
Rattus rattus **(black rat)**	60	*Capra hircus* (goats)	30
Mus musculus **(house mouse)**	49	*Myocastor coypus* (coypu)	21
Rattus norvegicus **(brown rat)**	48	*Oryctolagus cuniculus* (rabbits)	20
Felis catus **(cat)**	38	*Mustela vison* (American mink)	18
Sus scrofa **(feral pig)**	32	*Canis lupus familiaris* (dogs)	15

threatened vertebrate species (Spatz *et al.*, 2017). Among the orders richest in alien mammals, the highest per centage globally is for Rodentia (58 species, 25 per cent), Cetartiodactyla (49 species, 21 per cent), Carnivora (30 species, 13 per cent), Diprotodontia (28 species, 12 per cent) and Primates (26 species, 11 per cent) (Biancolini *et al.*, 2021). Some alien mammals such as *Rattus* spp. (rats), *Mus musculus* (house mouse) and *Felis catus* (cat) are so common that they are often not recognized as invasive alien species in mainland regions (Long, 2003; Loss & Marra, 2017), and thus are missing from lists of alien species. Several of these mammals have lived in close proximity to humans for a very long time resulting in long-lasting commensalisms (Puckett *et al.*, 2020) and in the spread of these species globally.

Many of the most widespread invasive alien mammals worldwide **(Table 2.5)**, such as feral domestic species and commensal stowaways, can exploit human-disturbed environments (Biancolini *et al.*, 2021; Long, 2003). On islands and in Australia, where invasive alien mammals are the main cause of extinction and native species declines (Courchamp *et al.*, 2003; Woinarski *et al.*, 2015), they are subject to many control and eradication measures (DIISE, 2020; H. P. Jones *et al.*, 2016; Parkes *et al.*, 2017; Russell *et al.*, 2015, 2016). Other notorious global invasive mammals include *Herpestes javanicus auropunctatus* (small Indian mongoose), *Oryctolagus cuniculus* (rabbits), *Lepus europaeus* (European hare), *Dama dama* (fallow deer), *Camelus dromedarius* (dromedary camel), *Ondatra zibethicus* (muskrat), *Mustela vison* (American mink), *Myocastor coypus* (coypu), Procyon lotor (raccoon), *Nyctereutes procyonoides* (raccoon dog), *Vulpes vulpes* (red fox), *Sus scrofa* (feral pig), *Capra hircus* (goats), *Ovis aries* (sheep), *Equus asinus* (donkeys), *Equus caballus* (horse), *Bos taurus* (cattle), and *Canis lupus familiaris* (dogs) (Biancolini *et al.*, 2021; Blackburn *et al.*, 2017; Clout & Russell, 2008; Long, 2003; Louppe *et al.*, 2020). Mammals are the most widespread group of invasive alien animal species in terms of the number of regions invaded **(Table 2.5)**.

2.3.1.2 Birds

Trends

Birds have been introduced for thousands of years, but a notable acceleration of introductions occurred in the mid-nineteenth century arising from increasing European colonial expansion and an acclimatization of alien species considered to be beneficial. The origins and introduction sites of alien birds during this period reflects the geography of colonialism, and the locations of former British colonies (E. E. Dyer, Cassey, *et al.*, 2017), and especially hotspots such as New Zealand, Australia, Hawaii, and the Mascarenes. In this period, alien species were mainly deliberately introduced for game or ornamentation such as gallinaceous birds, wildfowl, and pigeons (E. E. Dyer, Cassey, *et al.*, 2017). Other alien species were introduced for biocontrol of agricultural insect pests such as *Acridotheres tristis* (common myna) introduced from India to Mauritius to control *Nomadacris septemfasciata* (red locust) in 1762 (Shaanker & Ganeshaiah, 1992; Simmonds *et al.*, 1976).

Introduction rates again accelerated in the mid-twentieth century most likely due to increasing trade volumes, particularly for birds imported and exported for the pet trade **(Figure 2.8)**. Most recent introductions, reflected in the taxonomic composition, stem from unintentional escapes or releases from the caged bird trade. Commonly introduced species are parrots, estrildid finches, mynas, and starlings (E. E. Dyer, Cassey, *et al.*, 2017).

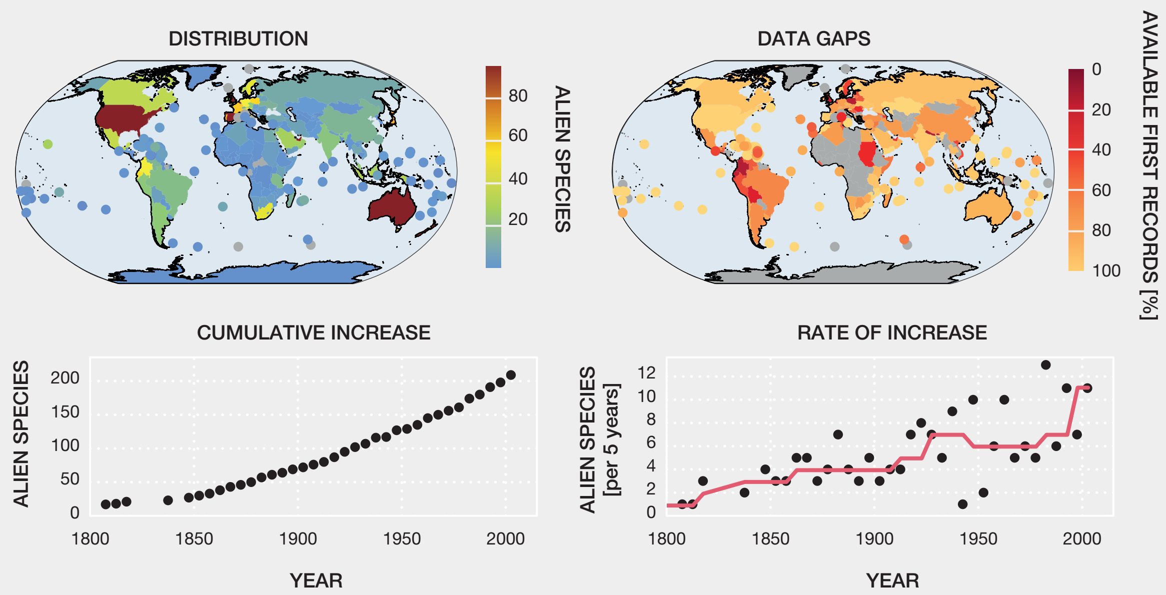

Figure 2 8 **Status, trends, and data gaps for established alien birds.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Status

Alien birds have been introduced to nearly all regions worldwide including many small islands (E. E. Dyer, Cassey, *et al.*, 2017; Evans, 2021). Global patterns of established alien bird species richness show relatively low numbers of alien birds in most parts of the world (though local numbers can be very high, e.g., more than 90 species in Hawaii), but very few regions without established alien bird species (Dawson *et al.*, 2017). E. E. Dyer, Cassey, *et al.* (2017) showed that colonization pressure (and to a smaller extent, distance from an historic port) was the key driver related to alien bird species richness, and that accounting for these factors, alien bird richness was also higher in areas with high native bird species richness. Thus, a range of environmental, life history, and anthropogenic factors determine areas with high alien bird richness.

A global analysis of historical data on bird introductions showed that environmental conditions at introduction sites are the primary determinants of successful establishment (Redding *et al.*, 2019). While climatic suitability is particularly important, the presence of other alien species can lead to an accumulation of alien species in "hotspots" potentially facilitating the establishment of additional species (termed "invasional meltdown"; **Glossary** and **Chapter 1**, **section 1.3.4**). Establishment of alien species is also more likely when extreme weather events do not occur in the decade following an introduction, suggesting that environmental stochasticity is important to the persistence of small populations (Redding *et al.*, 2019). Species-level traits, notably generalist species and founding population size, exert important secondary effects on success (Redding *et al.*, 2019). Generalist species are more likely to establish self-sustaining populations, as are species introduced in greater numbers (Cassey *et al.*, 2018; Redding *et al.*, 2019). Birds are strong dispersers, a trait that facilitates biological invasion success post-introduction (Cassey *et al.*, 2015). For example, of about 60 pairs of birds first introduced before the twentieth century to Central Park, New York City, *Sturnus vulgaris* (common starling) now numbers approximately 200 million individuals in the United States of America (Linz *et al.*, 2007).

Globally, particularly problematic invasive alien birds include *Anas platyrhynchos* (mallard), *Acridotheres tristis* (common myna), *Pycnonotus jocosus* (red-whiskered bulbul) (Martin-Albarracin *et al.*, 2015), *Nesoenas picturatus* (Madagascar turtle dove), *Pitangus sulphuratus* (great kiskadee), *Tyto novaehollandiae* (Australian masked owl), *Tyto alba* (barn owl), and *Bubo virginianus* (great horned owl) (Evans *et al.*, 2016). The 10 most widespread species are listed in **Table 2.6**.

Table 2 6 **Top 10 most widespread invasive alien bird species worldwide.**

The number of regions where the respective species has been recorded and classified as being invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien bird species, not impacts, which are covered in **Chapter 4** (see **section 2.1.4** for further details on data sources and processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Taxon	No. of regions	Taxon	No. of regions
Acridotheres tristis **(common myna)**	22	*Branta canadensis* (Canada goose)	9
Columba livia **(pigeons)**	20	*Alopochen aegyptiaca* (Egyptian goose)	8
Corvus splendens **(house crow)**	17	*Sturnus vulgaris* (common starling)	8
Passer domesticus **(house sparrow)**	14	*Myiopsitta monachus* (monk parakeet)	7
Psittacula krameri **(rose-ringed parakeet)**	13	*Phasianus colchicus* (common pheasant)	6

2.3.1.3 Fishes

Trends

Freshwater fish invasions are one of the best documented biological invasions among animal taxa with considerable information available on invasive alien fish traits, invaded regions, and invasion pathways (Bernery *et al.*, 2022). Information for marine fish invasions is much more fragmented (e.g., Arndt *et al.*, 2018; Vignon & Sasal, 2010). Globally, the number of invasive alien fishes accelerated in the twentieth century **(Figure 2.9)**. Although one might conclude that saturation has been reached based on the figure displaying the number of established alien species per five-year intervals, the lag between species introduction, reports of the introduction in the literature, and the cumulative numbers worldwide for this taxonomic group suggest that this is not the case (Seebens, Blackburn, *et al.*, 2017). Even though introductions of fish outside their natural ranges worldwide increased substantially at the onset of the industrial revolution, first records of alien fish introductions date back at least to the Roman Empire in Europe (first and second century; Balon, 1995).

Currently, the rate of newly established alien fish species is still very high, higher than for most other taxa (Seebens, Blackburn, *et al.*, 2017), partially explaining why fish are among the most widespread invasive alien taxonomic group (Gozlan, 2008). Globally, many fish species have been and are often still introduced intentionally, although unintentional introductions also occur. Due to widespread intentional introductions, alien freshwater fish species occur in all biogeographic regions (Leprieur *et al.*, 2008). Due to the compounding effects of increased global maritime transportation, canal construction, and climate change, the number of alien marine fish also rose dramatically in the twentieth and twenty-first centuries. These same three factors may also further promote biological invasions of fish in the future (Castellanos-Galindo *et al.*, 2020; Cohen, 2006; Muirhead *et al.*, 2015; Ruiz *et al.*, 2006).

Status

The most widespread alien fish species are listed in **Table 2.7** demonstrating the very high number of regions invaded by this group, second only to mammals in terms of distribution.

Dawson *et al.* (2017) showed that alien freshwater fish were distributed in six global biological invasion hotspots where established alien species constituted over 25 per cent of total species richness. When considering within country introductions, which are frequently not included in global analyses, the number of alien fishes increased for large countries such as Brazil, the People's Republic of China, and the United States (Vitule *et al.*, 2019). Pathways of fish biological invasions vary and include inter-oceanic canals, ballast water, intentional introductions for fishing or fisheries stocking, ornamental purposes, and escapes from aquaculture. For example, many alien populations of salmonids, tilapias, and carps originated from aquaculture escapes (Froese & Pauly, 2015). The Center for Food Safety reported about 26 million escaped fish worldwide between 1996 and 2012 (CFS, 2012). Similarly, D. Jackson *et al.* (2015) reported almost 9 million escapees in six European countries over a 3-year period. Estimates suggest that in Chile more than 1 million salmonids escape annually from the net pens of salmon farms (Sepúlveda *et al.*, 2013; Thorstad *et al.*, 2008). Marine waters are also inhabited by many alien fishes. The opening of the Suez Canal has enabled the migration of species from the Red Sea into the Mediterranean Sea (known as Lessepsian/Erythraean invasion), which has caused the influx of more

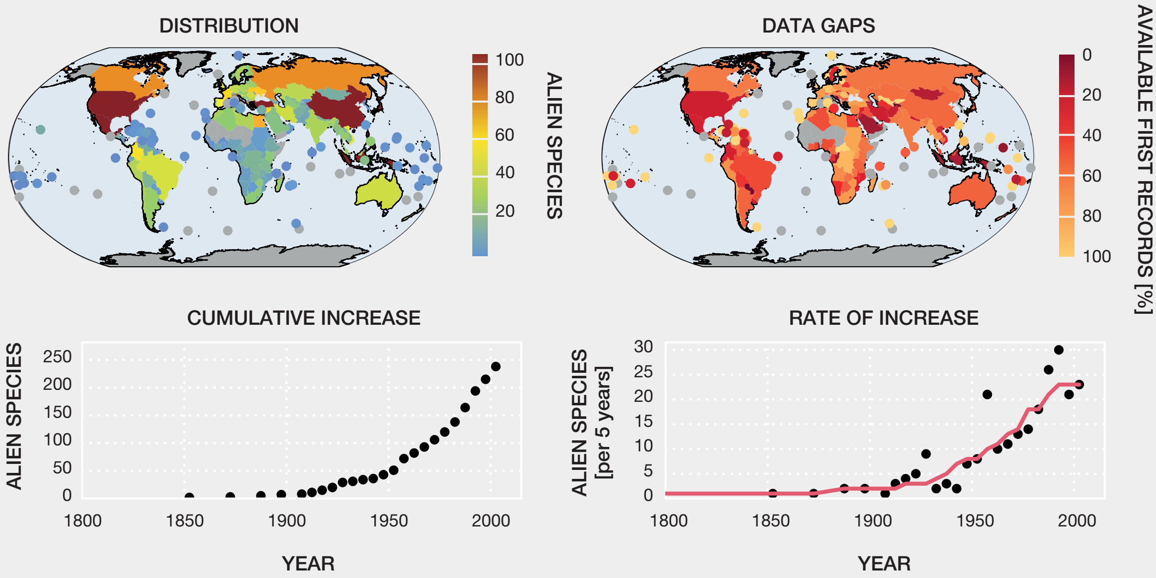

Figure 2 9 **Status, trends, and data gaps for established alien fishes.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted to aid visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as a running median (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Table 2 7 **Top 10 most widespread invasive alien fish species worldwide.**

The number of regions where the top 10 most widespread fishes have been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien species rather than impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Taxon	No. of regions	Taxon	No. of regions
***Cyprinus carpio* (common carp)**	43	*Poecilia reticulata* (guppy)	22
***Gambusia holbrooki* (eastern mosquitofish)**	42	*Pseudorasbora parva* (topmouth gudgeon)	22
***Oreochromis niloticus* (Nile tilapia)**	28	*Gambusia affinis* (western mosquitofish)	19
***Oreochromis mossambicus* (Mozambique tilapia)**	25	*Lepomis gibbosus* (pumpkinseed)	19
***Oncorhynchus mykiss* (rainbow trout)**	23	*Micropterus salmoides* (largemouth bass)	18

than 400 Indo-Pacific species into the Mediterranean Sea, including over 100 (118 by latest tally, unpublished) fish species (Bariche & Fricke, 2020; Çinar *et al.*, 2021; Galil *et al.*, 2021b), resulting in considerable changes to fish communities and fisheries, particularly in the Levant basin to date (Arndt *et al.*, 2018; Arndt & Schembri, 2015; Galil *et al.*, 2007). Both *Pterois volitans* (red lionfish) and *Pterois miles* (lionfish) have invaded large areas of the north-western Atlantic imposing large impacts on prey populations of native species and local fisheries (Côté *et al.*, 2013), and *Pterois miles* is now spreading within the Mediterranean Sea (Poursanidis *et al.*, 2020). Species of peacock basses

(genus *Cichla*), native to South America, have been introduced to tropical and sub-tropical regions worldwide for fisheries (Franco *et al.*, 2022).

2.3.1.4 Reptiles

Trends

The introduction of alien reptiles has a long history associated with the movement of humans and trade routes. For example, introduced species such as *Tarentola mauritanica* (common wall gecko) and *Vipera aspis* (asp viper) in the Mediterranean Basin can be traced back to the fourth century B.C. and the fifth century, respectively (Masseti & Zuffi, 2011; Mateo *et al.*, 2011; Pleguezuelos, 2002). Since 1800, the number of first records of alien reptiles globally has been rising steadily, accelerating since 1950 (Capinha *et al.*, 2017; Kraus, 2009). Similar trends have also been reported at local and regional scales (Krysko *et al.*, 2011, 2016; Mateo *et al.*, 2011; Perella & Behm, 2020; Powell *et al.*, 2011; Toomes *et al.*, 2020). Most alien reptile introductions through the end of the twentieth century were due to the unintentional transport of species as stowaways or contaminants (Kraus, 2009; Lever, 2003). This pathway remains important, but the pet trade has also emerged as a significant source of alien reptiles in recent decades (É. Fonseca *et al.*, 2019; Lockwood *et al.*, 2019; Perella & Behm, 2020; Stringham & Lockwood, 2018; Van Wilgen *et al.*, 2010).

Contemporary trends **(Figure 2.10)**, the expected increase in pet trade as a source of new species, and model-based projections of future distributions all indicate that both the number of alien reptiles and the number of invaded areas will continue to increase (Chapple *et al.*, 2016; da Rosa *et al.*, 2018; Filz *et al.*, 2018; Gippet & Bertelsmeier, 2021; X. Li *et al.*, 2016; X. Liu *et al.*, 2014; Seebens, Blackburn, *et al.*, 2017). Alien reptiles are fast becoming an important group of alien vertebrates alongside other taxa such as birds and mammals. In Australia, alien reptiles have been the dominant group of alien terrestrial vertebrates intercepted and detected at large since 1999 (Toomes *et al.*, 2020).

Status

Established populations of alien reptiles are found in all the IPBES regions except for the polar areas (Capinha *et al.*, 2017; Kraus, 2009). Islands and areas with relatively warm climates and high economic and human activity tend to host more alien reptiles than other places (Capinha *et al.*, 2017; É. Fonseca *et al.*, 2019; Moser *et al.*, 2018; Silva-Rocha *et al.*, 2019). Of the top five global hotspots for alien

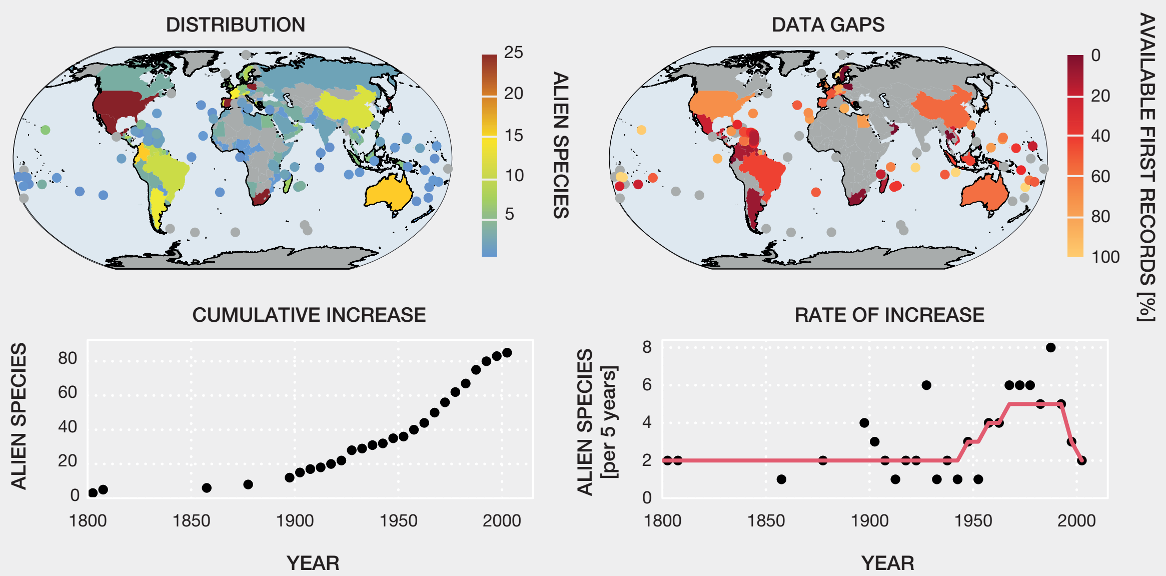

Figure 2 10 **Status, trends and data gaps for established alien reptiles.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization purposes and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Table 2 8 **Top 10 most widespread invasive alien reptile species worldwide.**

The table shows the number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table refers only to the distribution of invasive alien species, not their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Trachemys scripta elegans* (red-eared slider)**	15	*Chelydra serpentina* (common snapping turtle)	4
***Hemidactylus frenatus* (common house gecko)**	12	*Anolis cristatellus* (Puerto Rican crested anole)	4
***Hemidactylus mabouia* (tropical house gecko)**	12	*Anolis porcatus* (Cuban green anole)	3
***Iguana iguana* (iguana)**	8	*Hemidactylus turcicus* (Mediterranean house gecko)	3
***Anolis sagrei* (brown anole)**	5	*Pelodiscus sinensis* (Chinese soft-shelled turtle)	3

reptiles, the top three are in North America (Florida, Hawaii, and California), Europe (Balearic Islands, Spain), and Japan (Capinha *et al.*, 2017; Krysko *et al.*, 2011, 2016; Mateo *et al.*, 2011; Meshaka, 2011; Silva-Rocha *et al.*, 2015).

At least 198 reptile species belonging to three major reptile orders (Squamata, Crocodilia, and Testudines) have established alien populations worldwide (Capinha *et al.*, 2017). Of the top five most commonly established alien reptiles, four species (*Indotyphlops braminus* (brahminy blind snake), *Hemidactylus frenatus* (common house gecko), *Hemidactylus mabouia* (tropical house gecko), and *Hemidactylus turcicus* (Mediterranean house gecko)) have been transported unintentionally, and one (*Trachemys scripta* (pond slider)) is common in the pet trade (Capinha *et al.*, 2017; García-Díaz *et al.*, 2015; Kraus, 2009; Masin *et al.*, 2014). Some of the above species are among the 10 most widespread of all invasive alien reptiles worldwide **(Table 2.8)**. The establishment success and spread rates of alien reptiles are associated with high propagule pressure, the degree of climate matching between native and recipient regions, presence of congenerics, and high reproductive output (W. L. Allen *et al.*, 2017; Bomford *et al.*, 2009; X. Liu *et al.*, 2014; Mahoney *et al.*, 2014; Tingley *et al.*, 2016; Van Wilgen & Richardson, 2012). As examples, *Python bivittatus* (Burmese python) is spreading in the Florida Everglades, preying upon many species including the apex native predator *Alligator mississippiensis* (American alligator; Dorcas *et al.*, 2012). Invasive alien *Boiga irregularis* (brown tree snake) has reached iconic status as one of the most impactful invasive alien species worldwide. Fewer than 10 individuals were unintentionally introduced from the United States into the Pacific Island of Guam following World War II (Richmond *et al.*, 2015). This species has since colonized all habitats on Guam, from grasslands to forests, with peak densities as high as 10,000 individuals per km^2 (Rodda *et al.*, 1992). Several lesser known and potentially invasive alien reptiles are emerging including *Varanus niloticus* (Nile monitor) in Florida, *Lampropeltis getula* (common kingsnake) in the Canary Islands, *Boa constrictor* (boa constrictor) on Aruba, and several giant constrictor snakes in Puerto Rico (Reed & Kraus, 2010).

2.3.1.5 Amphibians

Trends

Alien amphibian introductions are not a new phenomenon. For instance, the introduction of *Bufotes balearicus* (Balearic green toad) to the Balearic Islands, Spain, is assumed to have occurred around the second century B.C. (Mateo *et al.*, 2011; Pleguezuelos, 2002). However, the accumulation of first records of alien amphibians shows a global rise since 1800 with a slightly more pronounced increase after the 1950s (Capinha *et al.*, 2017; Kraus, 2009, 2011). Similar patterns of relative increases in both the number of new alien species and the number of records of alien amphibians have been reported regionally and locally (Krysko *et al.*, 2011, 2016; Mateo *et al.*, 2011; Powell *et al.*, 2011; Toomes *et al.*, 2020). Nevertheless, the implementation of biosecurity and rapid response activities in countries such as New Zealand and Australia has likely prevented new introductions and establishment of alien amphibians (Chapple *et al.*, 2016; García-Díaz *et al.*, 2017; Toomes *et al.*, 2020). The United States appears to be an outlier in terms of new introductions; both the number of alien amphibian species

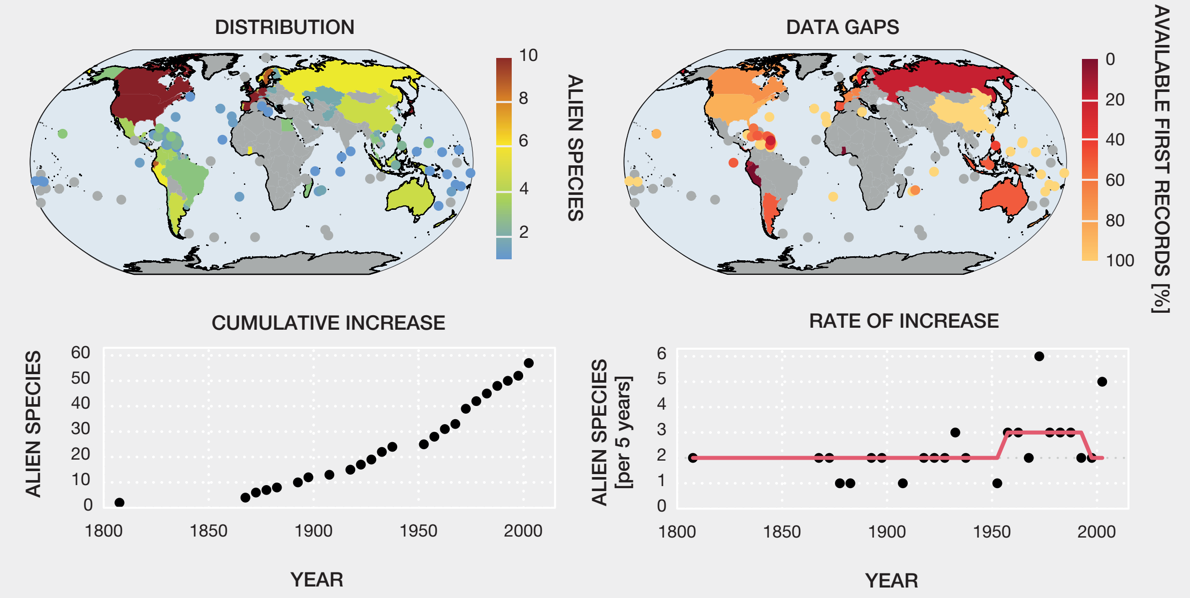

Figure 2 11 **Status, trends, and data gaps for established alien amphibians.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization purposes and do not indicate species numbers. The trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend line is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

reported annually and the number of records per year have remained relatively stable since around the mid-twentieth century (Mangiante *et al.*, 2018). It is important to note that in 2016 the United States Fish and Wildlife Service published an interim rule listing 201 salamander species as injurious wildlife under the Lacey Act to prevent the arrival of *Batrachochytrium salamandrivorans* (chytrid fungus) carried by alien species in the trade. Similarly, in 2017, Canada restricted salamander importation for the same reason (Yap *et al.*, 2017).

Status

Intentional and unintentional pathways are virtually equivalent contributors to the current distribution and status of alien amphibians worldwide, but their role varies by region and period (Kraus, 2009; Lever, 2003). For example, individuals of several toad species (family Bufonidae), such as *Rhinella marina* (cane toad) and *Sclerophrys gutturalis* (guttural toad), were deliberately released as biocontrol agents in the Indo-Pacific and Caribbean islands during the first half of the twentieth century (Kraus, 2009; Lever, 2003; Powell *et al.*, 2011; Shine, 2018; Telford *et al.*, 2019). More recently, *Duttaphrynus melanostictus* (Asian common toad) has been unintentionally transported to many areas in the Indo-Pacific region (Mo, 2017; Moore *et al.*, 2015; Tingley *et al.*, 2018; Vences *et al.*, 2017). The two most widespread alien amphibians in the world, *Lithobates catesbeianus* (American bullfrog) and *Rhinella marina*, have been introduced as a source of food and for biocontrol purposes, respectively (Capinha *et al.*, 2017; Kraus, 2009; X. Liu *et al.*, 2012, 2015; Shine, 2018). In Australia, almost twice the number of alien amphibians was found introduced through the pet trade compared to the stowaway pathway (71 and 38, respectively), yet the latter is a more important pathway when considering the total number of individuals moved rather than the number of species (García-Díaz & Cassey, 2014; Toomes *et al.*, 2020). Unintentional pathways are responsible for 12 out of 13 alien amphibians present in Guam (Christy, Clark, *et al.*, 2007). The pet trade is expected to remain a prominent source of new alien amphibian introductions in the near and medium-term (Lockwood *et al.*, 2019; Mohanty & Measey, 2019; Stringham & Lockwood, 2018).

The diversity of transport pathways responsible for the introduction of alien amphibians has resulted in established alien amphibian populations in all IPBES regions except for polar areas (**Figure 2.11**; Capinha *et al.*, 2017; Christy, Savidge, *et al.*, 2007; É. Fonseca *et al.*, 2019; García-Díaz

Table 2 9 **Top 10 most widespread invasive alien amphibian species worldwide.**

The table shows the number of regions where the respective species has been recorded and classified as being invasive based on GRIIS (Pagad *et al.*, 2022). Note that this table only refers to the distribution of invasive alien species rather than their impacts, which is covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Lithobates catesbeianus* (American bullfrog)**	24	*Pelophylax ridibundus* (Eurasian marsh frog)	3
***Rhinella marina* (cane toad)**	14	*Duttaphrynus melanostictus* (Asian common toad)	2
***Xenopus laevis* (African clawed frog)**	9	*Eleutherodactylus coqui* (Caribbean tree frog)	2
***Triturus carnifex* (Italian crested newt)**	3	*Eleutherodactylus planirostris* (greenhouse frog)	2
***Eleutherodactylus johnstonei* (whistling frog)**	3	*Andrias davidianus* (Chinese giant salamander)	1

& Cassey, 2014; Kraus, 2009; Measey *et al.*, 2017; Rago *et al.*, 2012; Tingley *et al.*, 2010). The United Kingdom, and California, Hawaii, and Puerto Rico (United States) are the top-four global hotspots of alien amphibians, each with more than five species established (Capinha *et al.*, 2017; Kraus, 2009; Powell *et al.*, 2011). Alien amphibian richness tends to be higher on islands and in places with high precipitation, high potential evapotranspiration, and high levels of economic activity (Capinha *et al.*, 2017; É. Fonseca *et al.*, 2019; Poessel *et al.*, 2012). High propagule pressure, the presence of congeneric species, life-history traits related to rapid growth and reproduction, and environmental similarity between the recipient and the native ranges are associated with the establishment success and invasion rates of alien amphibians (W. L. Allen *et al.*, 2017; Bomford *et al.*, 2009; Ferreira *et al.*, 2012; K. Li *et al.*, 2016; X. Liu *et al.*, 2014; Poessel *et al.*, 2012; Rago *et al.*, 2012; Tingley *et al.*, 2010, 2011; Van Wilgen & Richardson, 2012). It is interesting to note that many species native to Southern Africa have been introduced elsewhere, while few alien amphibians are reported for Southern Africa due to a very low trade involving these animals (Measey *et al.*, 2017).

The reported trajectories, combined with invasive alien amphibian niche shifts and the increase in pet trade, point to future increases in both the number of new alien amphibians and the number of regions occupied (Capinha *et al.*, 2017; Chapple *et al.*, 2016; da Rosa *et al.*, 2018; Mohanty *et al.*, 2021; Mohanty & Measey, 2019; Pili *et al.*, 2020; Toomes *et al.*, 2020). Additionally, invasion debts (i.e., the additional area an invasive alien species is likely occupy in the future; **Glossary**) mean that the accelerating trends in introductions described above could lead to established populations unless rapid response management actions are taken (Chapple *et al.*, 2016; M. J. Spear *et al.*, 2021).

Notorious invasive amphibians include *Rhinella marina* (cane toad), a large and toxic toad native to Mesoamerica and introduced worldwide into sugar cane producing regions to control beetles causing crop damage (Shanmuganathan *et al.*, 2010). *Xenopus laevis* (African clawed frog) is among the most commonly used laboratory animals (e.g., basic biology and formerly for pregnancy testing); many populations originating from laboratories have become invasive in regions with a Mediterranean climate. **Table 2.9** lists the 10 most widespread invasive alien amphibians and the number of regions each has invaded.

2.3.1.6 Insects

Trends

Since Insecta is the largest animal class it comes as no surprise that global numbers of alien insect species vastly exceed numbers for all other animal taxa combined by 1.7 times (Seebens, Blackburn, *et al.*, 2017). Yet, their biological invasions are still likely underreported as insects are less studied relative to other organisms such as vertebrates.

While there are a few rare documented cases of natural intercontinental insect spread (e.g., *via* wind) (Hoffmann & Courchamp, 2016), the long-distant spread of alien insects has risen steeply due to the facilitation by recent human activities (Gippet *et al.*, 2019; Meurisse *et al.*, 2019). Early exploration and colonial settlements facilitated the global range extension of several insect species, but higher rates of alien species establishment did not begin until approximately 1820 and lasted until 1914. This was followed by a second wave of accelerated establishment post-1960 (Bonnamour *et al.*, 2021). These periods coincided with the industrial revolution; increased global trade and travel facilitated accidental movement of insects with plants, plant products,

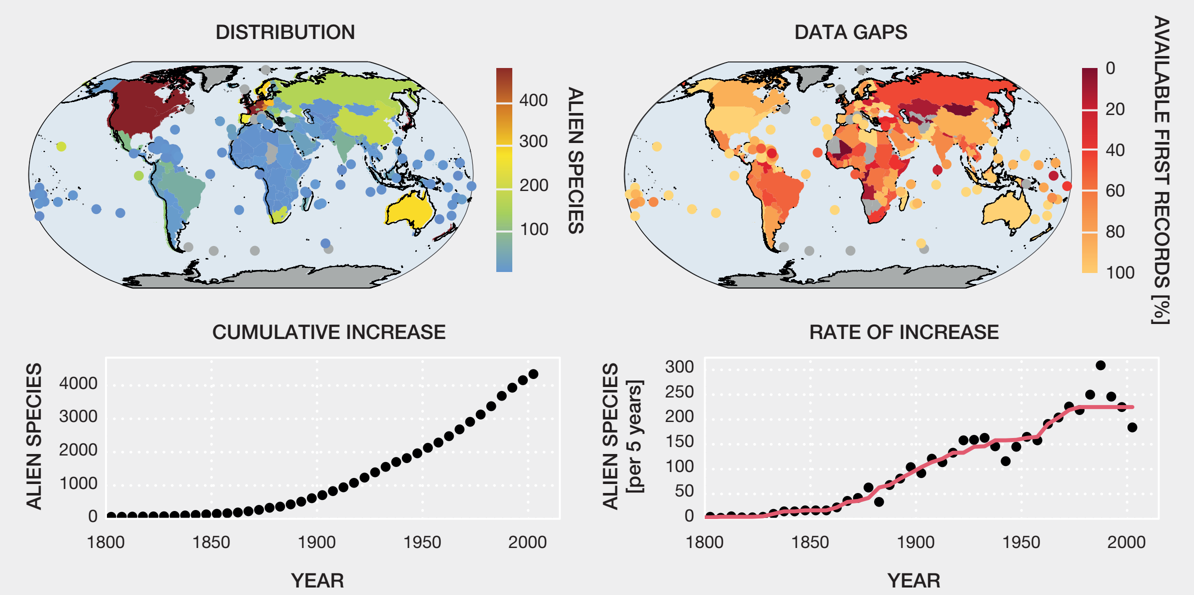

Figure 2 12 **Status, trends, and data gaps for established alien insects.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization purposes and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

general cargo, and baggage (Bertelsmeier *et al.*, 2017; Bonnamour *et al.*, 2021). Much of the global distribution of alien insects is driven by plant biological invasions (**Chapter 3**, **section 3.3.5.1**); many insects are dependent on individual plant species or genera, so establishment of alien plant species provides necessary resources that facilitate insect establishment (Liebhold *et al.*, 2018). Some evidence indicates that the recent implementation of biosecurity practices has reduced the proportion of imports contaminated with insects (Leung *et al.*, 2014; Liebhold & Griffin, 2016), but imports have also simultaneously and massively increased at the same time. While insects are such a large group that some specific variation may be masked, the resulting trend is a net increase. Indeed, as a group, they have even exponentially increased since the start of the nineteenth century, both in terms of cumulative numbers and number of established alien species per five-year intervals **(Figure 2.12)**, and still show no sign of saturation (Bonnamour *et al.*, 2021; Seebens, Blackburn, *et al.*, 2017). The continued increase of global trade and climate change will likely further accelerate for these easily transported and climate-sensitive organisms (Bellard, Thuiller, *et al.*, 2013). Additional factors could contribute to further spread (e.g., large infrastructure projects; Galil, Boero, Campbell, *et al.*, 2015; X. Liu *et al.*, 2019; Muirhead *et al.*, 2015) or establishment (e.g., industrial rearing of insects for food; Bang & Courchamp, 2021) of both existing and new invasive alien insects.

Status

Global estimates of the total number of alien insects are not available but likely exceed 10,000 species with more than 3,500 species established in North America alone (Yamanaka *et al.*, 2015). Actual numbers are likely much higher since many established species remain undiscovered or unreported. Global hotspots of insect biological invasions appear to be related to historical patterns of urbanization and industrialization (Branco *et al.*, 2019; Huang *et al.*, 2011) and the transport of species between Europe, East Asia, and North America reflecting trade and travel patterns (Kenis *et al.*, 2007; Mattson *et al.*, 2007). As global connectivity increases, regions such as Africa and South America are likely to be increasingly important as both recipients and donors of invasive alien insects.

Many invasive alien insects are highly problematic around the world, with coleopterans, lepidopterans, dipterans, and hymenopterans being among the most notorious (e.g., Kenis *et al.*, 2009). For example, alien ant species are often considered among the worst invasive alien species (Holway *et al.*, 2002; Pyšek *et al.*, 2008). Three ants are among the 10 most widespread invasive insects **(Table 2.10)** and five are among the "100 of the world's worst invasive alien species", the only family to have so many species listed. Ants are easily transported by humans because of their generalist nesting habits and their small size

(Wetterer *et al.*, 2009). When intercepted at ports of entry, alien ant species are frequently detected on commercial ornamental plants (Lester, 2005; Suarez *et al.*, 2005; Ward *et al.*, 2006). Globally, more than 200 species have established populations outside their native distributions (Wetterer *et al.*, 2009), but over 600 species have likely been introduced outside their native ranges (Miravete *et al.*, 2014). This makes ants the most represented insect family and particularly notorious ant species include *Linepithema humile* (Argentine ant), *Anoplolepis gracilipes* (yellow crazy ant), *Wasmannia auropunctata* (little fire ant), *Solenopsis invicta* (red imported fire ant), and *Pheidole megacephala* (big-headed ant). In addition, a recent study predicted that 13 other species with similar ecological traits could also become invasive should they be introduced outside their native ranges (Fournier *et al.*, 2019). To date, few studies are available on the biology and ecology of these invasive alien ants, except for *Linepithema humile* and *Solenopsis invicta* (Bertelsmeier *et al.*, 2016; Pyšek *et al.*, 2008). These two ant species from South America have invaded many countries by separate multiple introductions from their native ranges and subsequent secondary spread from invaded ranges (Ascunce *et al.*, 2011; Giraud *et al.*, 2002). Secondary introduction seems to be common for ants: 76 per cent of interception events of alien ants at the border of the United States and 88 per cent of those intercepted at the New Zealand border did not come from their country of origin but from previously invaded countries (Bertelsmeier *et al.*, 2018).

Many alien insects are invasive in most parts of the world making it difficult to define the most important while remaining concise, but the 10 most widespread species provide good examples **(Table 2.10)**. *Ceratitis capitata* (Mediterranean fruit fly) and *Bemisia tabaci* (tobacco whitefly) affect agriculture in numerous countries, while insect-borne diseases are spread by the invasions of several mosquito species, such as *Aedes albopictus* (Asian tiger mosquito), *Aedes aegypti* (yellow fever mosquito), and *Anopheles quadrimaculatus* (common malaria mosquito). *Harmonia axyridis* (harlequin ladybird) was introduced to North America and Europe to control aphids, subsequently leading to the decline of native ladybirds through predation (Roy *et al.*, 2012). *Icerya purchasi* (cottony cushion scale) is found in most regions, where it feeds on more than 80 families of woody plants, particularly citrus crops. *Brontispa longissima* (coconut hispine beetle) feeds on young leaves of coconut palms throughout the Pacific region. *Bemisia tabaci* thrives in tropical and subtropical (and to a lesser degree temperate) regions, where it feeds on many plants but also facilitates the spread of plant viruses. Although not among the 10 most widespread, some other insects are among the best known of all invasive alien species. For example, North American forests have been deeply damaged by the invasions of *Agrilus planipennis* (emerald ash borer; Herms & McCullough, 2014; Poland & McCullough, 2006; Valenta *et al.*, 2017), *Anoplophora glabripennis* (Asian longhorned beetle; Dodds & Orwig, 2011; Kappel *et al.*, 2017; Nowak *et al.*, 2001), and *Lymantria dispar* (gypsy moth; C. B. Davidson *et al.*, 1999; Tobin *et al.*, 2012). *Drosophila suzukii* (spotted wing drosophila), a vinegar fly of Asian origin, has emerged as a devastating pest of small and stone fruits throughout North America, Europe and South America (L. A. dos Santos *et al.*, 2017). *Coptotermes formosanus* (Formosan subterranean termite) affects infrastructure and *Trogoderma granarium* (khapra beetle) destroys grain and seed reserves throughout the world. It is noteworthy that bees (Apis (honey bee), *Bombus* (bumble bee) or *Megachile* (leaf-cutter bees), among others; e.g., Bartomeus *et al.*, 2013; Goulson, 2003; Morales *et al.*, 2017) and wasps (*Vespa, Vespula*, gall and parasitoid wasps, among others; e.g.,

Table 2 10 **Top 10 most widespread invasive alien insect species worldwide.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table refers only to the distribution of invasive alien species, not their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Icerya purchasi* (cottony cushion scale)**	29	*Harmonia axyridis* (harlequin ladybird)	14
Tapinoma melanocephalum (ghost ant)	28	*Ceratitis capitata* (Mediterranean fruit fly)	14
Pheidole megacephala (big-headed ant)	24	*Brontispa longissima* (coconut hispine beetle)	13
Aedes albopictus (Asian tiger mosquito)	24	*Bemisia tabaci* (tobacco whitefly)	13
Solenopsis geminata (tropical fire ant)	19	*Cameraria ohridella* (horsechestnut leafminer)	13

Beggs *et al.*, 2011; Lester & Beggs, 2019) excepting *Apis mellifera scutellata* (Africanized bee), hybrid of several European honey bee subspecies and the East African honey bee, are the source of considerable revenue and rarely viewed as invasive despite outcompeting native pollinators (IPBES, 2016; Moritz *et al.*, 2005).

2.3.1.7 Arachnids

Trends

The number of recorded alien spiders has been increasing continuously (**Figure 2.13**; Nentwig, 2015; Seebens, Blackburn, *et al.*, 2017). An accelerated increase is observed after 1950 similar to those in many other invertebrate groups and likely as a consequence of increasing global trade and transport. In addition to the total number of alien spiders, the rate of annual new records has increased until the present reaching about 30 new records per five years (i.e., 6 new records annually; **Figure 2.13**).

Status

Worldwide, 285 alien spider species (0.57 per cent of all described spider species) have been recorded outside of their native range. Most alien spiders are known from only a few records, from a few regions, but some species are so widespread that they are alien to several continents **(Table 2.11)**. The 28 most widespread species (10 per cent of all alien spiders) are known from more than 30 invaded regions (often from all or most continents) and represent 50 per cent of all records. Major trade routes, at least past routes, connect areas of origin to invaded regions: 29 per cent of all globally spread spider species are native to Europe (while Europe is home to only 10 per cent of all spider species), 25 per cent from the Americas, 20 per cent from Asia, 17 per cent from Africa, 10 per cent from Australasia and the Pacific. Most spiders alien to Europe were unintentionally introduced either as stowaways, in or on transport vectors (i.e., the physical means or agent that transports a species; **Glossary**), or as contaminants (Nentwig, 2015). Horticulture is a major source of introduced spiders, followed by fruit and vegetable shipments, containers, and packaging materials. Imported classic cars and used sport cars often contained *Latrodectus mactans* (black widow spider) and cocoons in high numbers (Van Keer, 2010). For many areas in the world, no reliable species inventories are available. The top 10 most widespread invasive alien arachnids as recorded by GRIIS are shown in **Table 2.12**.

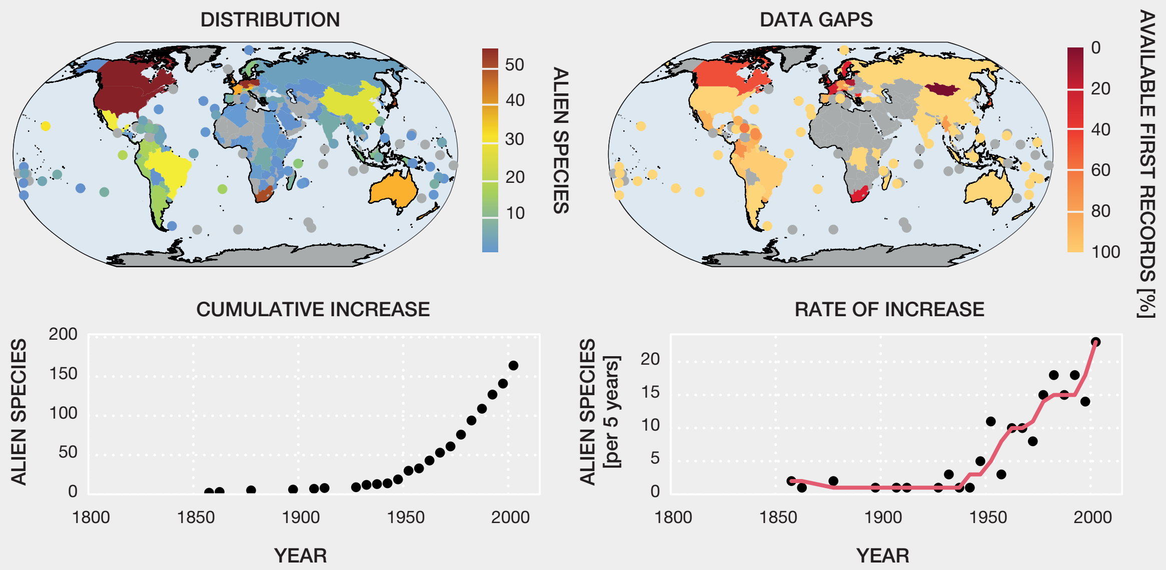

Figure 2 13 **Status, trends, and data gaps for established alien arachnids.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Table 2 11 **The most common established alien spider families and species.**

Based on 12 arachnid families with the most widely distributed established alien species, this family-wise presentation is of those species known to occur in more than 30 regions outside their native ranges. Families are ordered alphabetically, species according to frequency in the invaded area. Data from the World Spider Catalog (2017).

Family	No. of established alien species	Most widespread species	Alien range
Agelenidae **(funnel web spiders)**	8	*Tegenaria domestica* *Eratigena agrestis*	Europe Europe
Araneidae **(orb weavers)**	23	*Neoscona nautica* *Argiope trifasciata*	Pacific North America
Cheiracanthiidae **(yellow sac spiders)**	3	*Cheiracanthium mildei*	Europe
Dysderidae **(woodlouse hunters)**	2	*Dysdera crocata*	Pacific Europe North America
Oonopidae **(goblin spiders)**	19	*Triaeris stenaspis* *Brignolia parumpunctata* *Ischnothyreus peltifer* *Opopaea concolor*	Africa Tropical Asia Tropical Asia Africa
Pholcidae **(daddy-long-legs)**	15	*Pholcus phalangioides* *Micropholcus fauroti* *Artema atlanta* *Smeringopus pallidus* *Spermophora senoculata*	Temperate Asia Temperate Asia Africa Africa Temperate Asia
Salticidae **(jumping spiders)**	34	*Plexippus paykulli* *Hasarius adansoni* *Menemerus bivittatus*	Africa Africa Africa
Scytodidae **(spitting spiders)**	8	*Scytodes thoracica*	Europe
Oecobiidae **(disk web spiders)**	9	*Oecobius navus*	Africa
Sicariidae **(six-eyed spiders)**	1	*Loxosceles rufescens*	North America Europe Australia Asia
Sparassidae **(giant crab spiders)**	3	*Heteropoda venatoria*	Tropical Asia
Theridiidae **(cobweb or combfooted spiders)**	47	*Parasteatoda tepidariorum* *Steatoda grossa* *Steatoda triangulosa* *Latrodectus geometricus*	South America Europe Europe Africa

Table 2 12 **Top 10 most widespread invasive alien arachnids worldwide.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien species rather than their impacts which is covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
Raoiella indica **(red palm mite)**	7	*Steatoda nobilis* (false widow spider)	2
Opilio canestrinii **(harvestman)**	3	*Tetranychus urticae* (two-spotted spider mite)	2
Varroa destructor **(Varroa mite)**	3	*Aceria litchii* (litchi gall mite)	1
Latrodectus geometricus **(brown widow spider)**	2	*Aceria tristriata* (walnut leaf gall mite)	1
Mermessus trilobatus **(trilobate dwarf weaver)**	2	*Aculops lycopersici* (tomato russet mite)	1

2.3.1.8 Molluscs

Trends

Overall, molluscs have mostly been introduced unintentionally with numbers of introductions starting to increase at the end of 1800s **(Figure 2.14)**. Similar to crustaceans, marine species introductions started when transoceanic voyages began around 1500 but were rarely documented (Carlton, 1999b). During the second half of the twentieth century, increases in shipping, aquaculture, and the aquarium trade facilitated the introductions of both marine and freshwater molluscs (Carlton, 1999a; Cianfanelli *et al.*, 2016; Cowie, 2005; Darrigran *et al.*, 2020; De Silva, 2012; X. Guo, 2009; Katsanevakis *et al.*, 2013; Ojaveer *et al.*, 2018; R. Sousa *et al.*, 2014). A similar pattern is observed for terrestrial molluscs; they are almost exclusively moved as contaminants through agriculture and horticulture and their introductions began in ancient times (Herbert, 2010). Since 1600, European colonists have introduced many species to new areas (Herbert, 2010). With the increasing trade, introductions rates grew from the 1950s onward (Cowie, 2005; Herbert, 2010; Hutchinson *et al.*, 2014).

Status

Established alien molluscs have been reported from all over the world (Capinha *et al.*, 2015; R. Sousa *et al.*, 2009). However, despite their status as widespread alien species and extensive work by malacologists in terrestrial and marine ecosystems **(Figure 2.14)** their distribution and spread has received comparatively little attention except for species such as *Dreissena* spp. (zebra and quagga mussels), *Corbicula fluminea* (Asian clam), and *Magallana gigas* (Pacific oyster) (Dölle & Kurzmann, 2020; Orlova *et al.*, 2005; Ruesink *et al.*, 2005; A. Sousa *et al.*, 2009; Strayer *et al.*, 2019). For bivalves, R. Sousa *et al.* (2009) listed examples of 35 established alien species in marine and freshwater systems of all continents, 24 of which have sufficient information about distribution or effects reported. However, the number of established alien bivalves is likely much higher than reported. Recently, *Mytilus* cf. *platensis* (mussel) was discovered in Antarctic waters (Cárdenas *et al.*, 2020), further demonstrating that molluscs are transported in intercontinental transfers. Invasive bivalves often occur at very high densities becoming a major proportion of the benthic fauna (e.g., *Arcuatula senhousia* (Asian date mussel; Crooks & Khim, 1999), *Mytilus galloprovincialis* (Mediterranean mussel; Branch & Steffani,

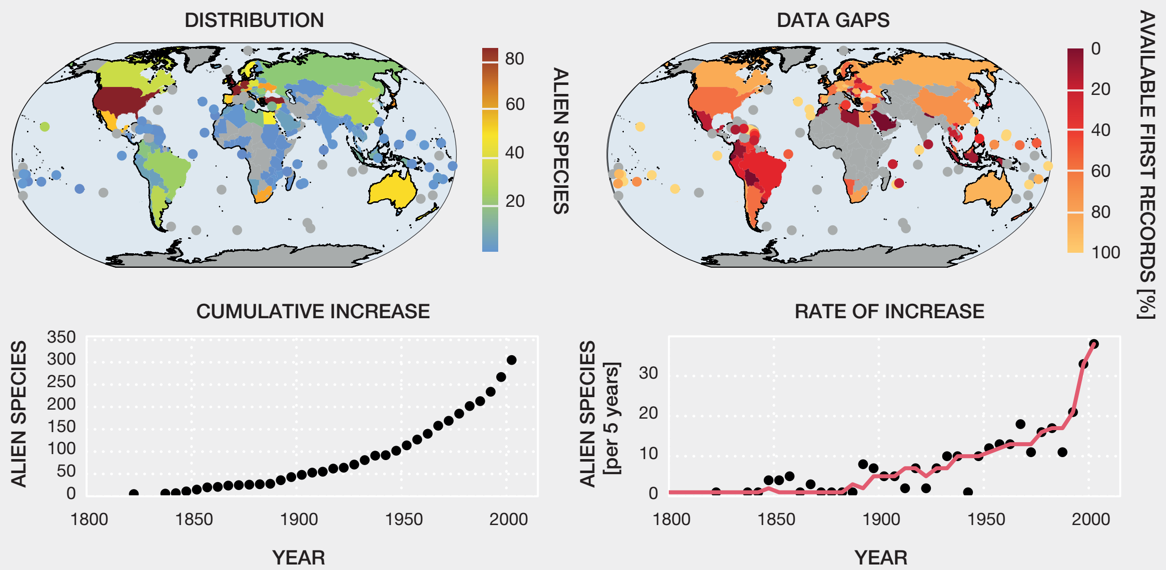

Figure 2 14 **Status, trends, and data gaps for established alien molluscs.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. The trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

2004), *Limnoperna fortunei* (golden mussel; Boltovskoy *et al.*, 2006), *Perna viridis* (Asian green mussel; Rajagopal *et al.*, 2006), and *Ensis leei* (American jack-knife clam; Raybaud *et al.*, 2015)).

Marine bivalves (oysters, mussels, clams) have long been widely introduced for cultivation and harvesting in many regions of the world. Some were introduced to replace depleted or diseased stocks of commercially valuable indigenous species, for example, *Magallana gigas* (Pacific oyster) and *Ruditapes philippinarum* (Japanese carpet shell) in Europe to diversify local marine farming, and *Mytilus edulis* (common blue mussel) in Canada and China (Tang *et al.*, 2002). These alien species cause negative impacts in their introduced habitats by forming reefs on hard and soft bottoms and effecting large structural changes in littoral communities (**Chapter 4**, **section 4.3.2.3**).

Though of small size, some invasive alien molluscs attain high densities and cause remarkable impacts. *Littorina littorea* (common periwinkle) occurs at densities of up to 600 individuals per m^2 (Carlson *et al.*, 2006), reduces algal canopies, and controls rocky intertidal community structure and species diversity (Bertness, 1984; Lubchenco, 1978; Petraitis, 1987; Yamada & Mansour, 1987). *Crepidula fornicata* (American slipper limpet) was introduced from the North American Atlantic coast to the Pacific coast and to Europe with *Crassostrea virginica* (eastern oyster). It forms dense conglomerations of live specimens, shells and pseudofaeces, transforming the physical and chemical composition of the sediment, which adversely affects the endobenthic community and reduces the area of flatfish habitat. When it fouls *Mytilus edulis* (common blue mussel), *Crepidula fornicata* increases mussel mortality by four to eight times, but also reduces mussel predation by *Asterias rubens* (common starfish; Blanchard, 2009; Kostecki *et al.*, 2011; Thieltges, 2005a, 2005b). The easternmost Mediterranean is the region with the highest reported number of marine alien molluscs (over 160 species along 180 kms of Israeli and Palestine coast alone), most introduced through the Suez Canal (Galil *et al.*, 2021b).

Alien snails and slugs have become established in most parts of the world, including on many islands. For example, 38 alien terrestrial snails and slugs are established in Hawaii (Cowie *et al.*, 2008). Cowie *et al.* (2009) listed 46 species spanning 18 families for priority quarantine from the United States. *Lissachatina fulica* (giant African land snail) is one of the largest land snails in the world, reaching up to 19 cm in length, and is recognized as one of the world's most damaging invasive alien species because of its omnivorous nature and because it is a vector of at least two human diseases (W. M. Meyer *et al.*, 2008; **Chapter 4**, **section 4.5.1.3**). *Euglandina rosea* (rosy predator snail) was originally introduced to control *Lissachatina fulica*. Not only did it fail to control it, but *Euglandina rosea* caused the extinction of many endemic snails on the islands of Hawaii, Tahiti, Moorea, and other Pacific islands (Davis-Berg, 2012; **Chapter 4**, **section 4.3.1**). Other widespread alien species include *Pomacea canaliculata* (golden apple snail; Q.-Q. Yang *et al.*, 2018), *Arion ater* (european black slug; Zemanova *et al.*, 2018), *Cepaea nemoralis* (grove snail), *Cornu aspersum* (common garden snail), *Limax maximus* (leopard slug), *Cernuella virgata* (vineyard snail), *Theba pisana* (white garden snail) and *Arion vulgaris* (Spanish slug). **Table 2.13** lists the 10 most widespread alien mollusc species invasive in most regions.

Table 2 13 **Top 10 most widespread invasive alien mollusc species worldwide.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien mollusc species rather than their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Lissachatina fulica* (giant African land snail)**	31	*Pomacea canaliculata* (golden apple snail)	13
***Corbicula fluminea* (Asian clam)**	22	*Arcuatula senhousia* (Asian date mussel)	10
***Dreissena polymorpha* (zebra mussel)**	20	*Melanoides tuberculata* (red-rimmed melania)	10
***Magallana gigas* (Pacific oyster)**	15	*Corbicula fluminalis* (Asian clam)	9
***Potamopyrgus antipodarum* (New Zealand mudsnail)**	15	*Dreissena rostriformis bugensis* (quagga mussel)	9

2.3.1.9 Crustaceans

Trends

Unintentional introductions of marine crustaceans probably began in the 1500s when transoceanic voyages were first undertaken (Carlton, 2011), but no data are available. The first records of alien crustaceans were reported between the 1800s and the beginning of 1900s (Carlton, 2011; **Figure 2.15**). Like those of other alien marine species, crustacean introductions have risen in recent decades due to increased shipping, fisheries, aquaculture, and aquarium trade (Fernández de Alaiza García Madrigal *et al.*, 2018; Hänfling *et al.*, 2011; Katsanevakis *et al.*, 2013; Ojaveer *et al.*, 2018). For example, the Suez Canal allowed the entry of alien crustaceans into the Mediterranean Sea for the entire twentieth century with an increase from 1990 facilitated by climate warming (Galil, 2011). The unintentional introduction of freshwater species started with global shipping and the construction of artificial canals (e.g., in Central and Western Europe), increasing after the 1950s. Overall, crustaceans were one of the most frequently introduced groups in recent decades in the Baltic Sea, California Bay, and the Laurentian Great Lakes (Hänfling *et al.*, 2011). On the other hand, crayfish have been intentionally introduced as a food source since the end of 1800s (Hänfling *et al.*, 2011), but global increases of crayfish production starting in the 1970s boosted introductions (Haubrock *et al.*, 2021; Lodge *et al.*, 2012).

Status

Crustaceans are frequently found among lists of marine and freshwater alien species (Galil *et al.*, 2011; Hänfling *et al.*, 2011; Simões *et al.*, 2021). As an example, the Mediterranean, North East Atlantic, Black and Baltic Seas host some of the highest species numbers, with 1,411 established alien species reported (Tsiamis *et al.*, 2018), a noteworthy proportion of which includes crustaceans (Tsiamis *et al.*, 2020). Owing to human activities, many marine crustacean species have achieved global distributions (e.g., barnacles *Balanus glandula* (Kerckhof *et al.*, 2018), *Amphibalanus improvisus* (bay barnacle), and *Amphibalanus eburneus* (ivory barnacle); isopods *Synidotea laevidorsalis* (J. W. Chapman & Carlton, 1991) and *Ianiropsis serricaudis*; amphipod *Caprella mutica* (Japanese skeleton shrimp); shrimp *Palaemon macrodactylus* (oriental shrimp); additional shrimp and many crab species; many copepods and mysids; and several more).

Hemigrapsus sanguineus (Asian shore crab) is now the dominant crab in rocky intertidal habitats along much of the north-eastern coast of the United States and the

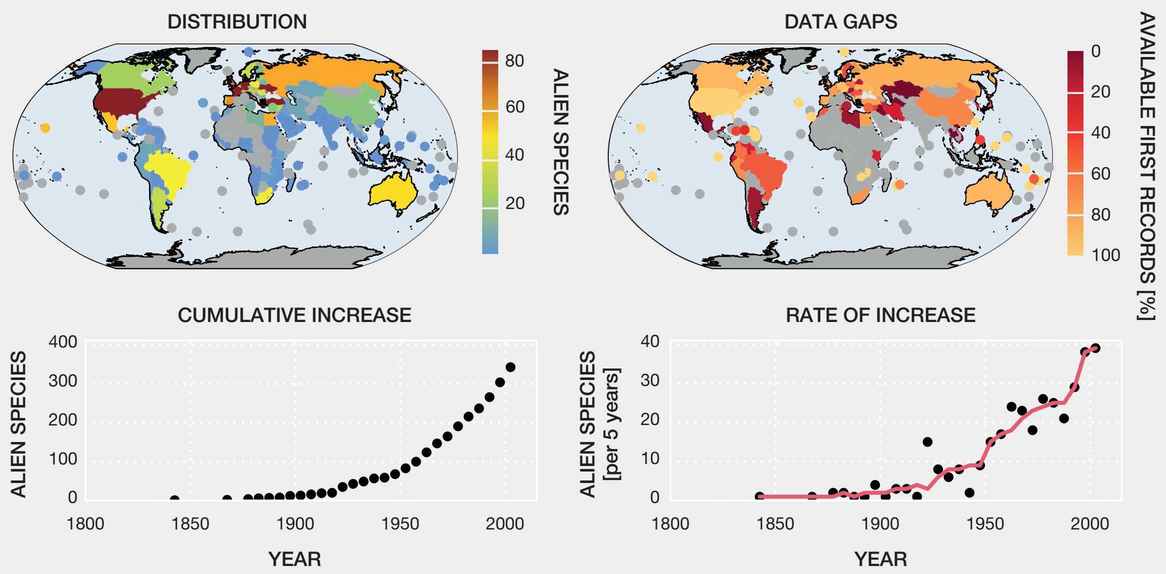

Figure 2 15 **Status, trends, and data gaps for established alien crustaceans.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. The trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend line is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Table 2 14 **Top 10 most widespread invasive alien crustacean species worldwide.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien crustacean species rather than their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Pacifastacus leniusculus* (American signal crayfish)**	19	*Dikerogammarus villosus* (killer shrimp)	12
***Procambarus clarkii* (red swamp crawfish)**	19	*Cherax quadricarinatus* (redclaw crayfish)	11
***Amphibalanus improvisus* (bay barnacle)**	17	*Chelicorophium curvispinum* (Caspian mud shrimp)	10
***Faxonius limosus* (spiny-cheek crayfish)**	14	*Cercopagis pengoi* (fishhook waterflea)	8
***Eriocheir sinensis* (Chinese mitten crab)**	12	*Macrobrachium rosenbergii* (giant freshwater prawn)	7

European Atlantic coast where it has been introduced and displaces resident crab species (Blakeslee *et al.*, 2017; Epifanio, 2013). The literature on the Asian shore crab is limited in comparison to that of better-known global marine invasive established crabs like *Carcinus maenas* (European shore crab), *Carcinus aestuarii* (Mediterranean green crab) (Cosham *et al.*, 2016; Leignel *et al.*, 2014), and *Eriocheir sinensis* (Chinese mitten crab; Dittel & Epifanio, 2009). **Table 2.14** lists the 10 most widespread invasive alien crustacean species and the number of regions each has invaded.

Crustaceans also comprise major proportions of alien animals established in large freshwater ecosystems; their rate of discovery, along with that of other freshwater invertebrates, is increasing in these habitats (Ricciardi, 2015). According to Gherardi (2010), 28 crayfish species have been introduced into a new biogeographic region and/or translocated within their native biogeographic region. In Europe, most crayfish species are alien (at least 10 alien, five native), with significantly higher abundances and severe impacts caused by alien crayfish, especially the transmission of crayfish plague, a disease lethal to native species (Kouba *et al.*, 2014; **Chapter 4**, **section 4.3.2.2**). There is increasing recognition of their severe impacts, notably the displacement of native species (Gherardi, Aquiloni, *et al.*, 2011; South *et al.*, 2020). In Africa, five out of nine introduced crayfish species established populations in at least six countries, causing substantial ecological and economic damage (Madzivanzira *et al.*, 2021). Genetic divergence between European and North American lineages of freshwater cladocerans suggests that the current rate of invasion by European species in North America is ca. 50,000 times higher than prehistoric levels (Hebert & Cristescu, 2002). Invasions of the Laurentian Great Lakes **(Box 2.11)** by two cladocerans, *Cercopagis pengoi* (fishhook waterflea), and *Bythotrephes longimanus* (spiny waterflea), have caused concern for freshwater biodiversity and regional fisheries (Pichlová-Ptáčníková & Vanderploeg, 2009). *Dikerogammarus villosus* (killer shrimp) is a physiologically tolerant and adaptable amphipod of Ponto-Caspian origin that has colonized most of the major European inland waterways in only two decades, replacing many local amphipod species. Its continued range expansion, as well as its potential to reach freshwaters of other continents (particularly North America and its Great Lakes), is a major conservation concern (Rewicz *et al.*, 2014). *Hemimysis anomala* (bloody-red shrimp) was one of several Ponto-Caspian species to invade the Great Lakes in recent decades through transoceanic shipping (Audzijonyte *et al.*, 2007).

2.3.1.10 Other invertebrates

Other invertebrates cover those invertebrate species that are not addressed in previous sections and include the phyla Acanthocephala, Annelida, Brachiopoda, Bryozoa, Chaetognatha, Cnidaria, Ctenophora, Echinodermata, Kamptozoa, Nematoda, Nemertea, Onychophora, Phoronida, Platyhelminthes, Porifera, Rotifera, Sipuncula and Xenacoelomorpha.

Trends

There is a paucity of data on molluscs, and crustaceans, but there is nothing to suggest that the trends for these animals differ from the better documented groups. In fact, data on the trends in both cumulative numbers and number of established alien species per five-year intervals show that animals other than the aforementioned vertebrates and invertebrates follow the same dramatic global increases since ca. 1850 **(Figure 2.16)**. For example, jellyfish populations appear to be increasing post-1950 in coastal ecosystems worldwide, mostly due to increasing populations of invasive alien species (Brotz *et al.*, 2012; importantly, note that Brotz *et al.* (2012) defined "jellyfish" as including three separate phyla of marine invertebrates – Cnidaria, Ctenophora, and Chordata). The increase has accelerated in recent decades and climate change is likely playing a role in facilitating increased survival and growth, and access to previously unfavourable waters. The depletion of predators and food competitors due to overfishing was also important (A. J. Richardson *et al.*, 2009). Notably, several comb jelly species (ctenophores) often survive ballast-water exchange, and their populations have been found to expand in over-fished areas that provide favorable conditions (Daskalov *et al.*, 2007). The invasion of the Black, Caspian, Baltic, and North Seas by the comb jelly *Mnemiopsis leidyi* (sea walnut) in the recent decades is a good illustration (Boersma *et al.*, 2007; Daskalov & Mamedov, 2007; Haslob *et al.*, 2007; Zaitsev, 1992). The increase of invasive alien jellyfish and comb jellies is predicted to continue accelerating (A. J. Richardson *et al.*, 2009). Other marine species, such as *Anemonia alicemartinae* (sea anemone), are considered invasive along the coast of Chile, and historical records show a rapid expansion towards the south, extending its distribution (Castilla *et al.*, 2005; Castilla & Neill, 2009; Häussermann & Försterra, 2001).

Status

Comprehensive studies for invertebrates, other than those reported above, are often lacking and detailed knowledge is usually available for only a few species. *Asterias amurensis* (northern Pacific seastar) is considered one of the most serious marine pests in Australia (MPSC, 2018). The same concern arises for *Centrostephanus rodgersii* (long-spined sea urchin). Its invasion from mainland Australia to Tasmania has already caused ecosystem shifts from kelp-dominated to a macroalgal-free habitat resulting in localized losses of about 150 taxa that associate with seaweed beds (Ling *et al.*, 2009). Among ctenophores, a prominent representative is the previously mentioned *Mnemiopsis leidyi* (sea walnut), first introduced from the North American east coast to the Black Sea in ship ballast water. The species subsequently spread throughout the Ponto-Caspian basin and the Mediterranean Sea, ultimately spreading across most

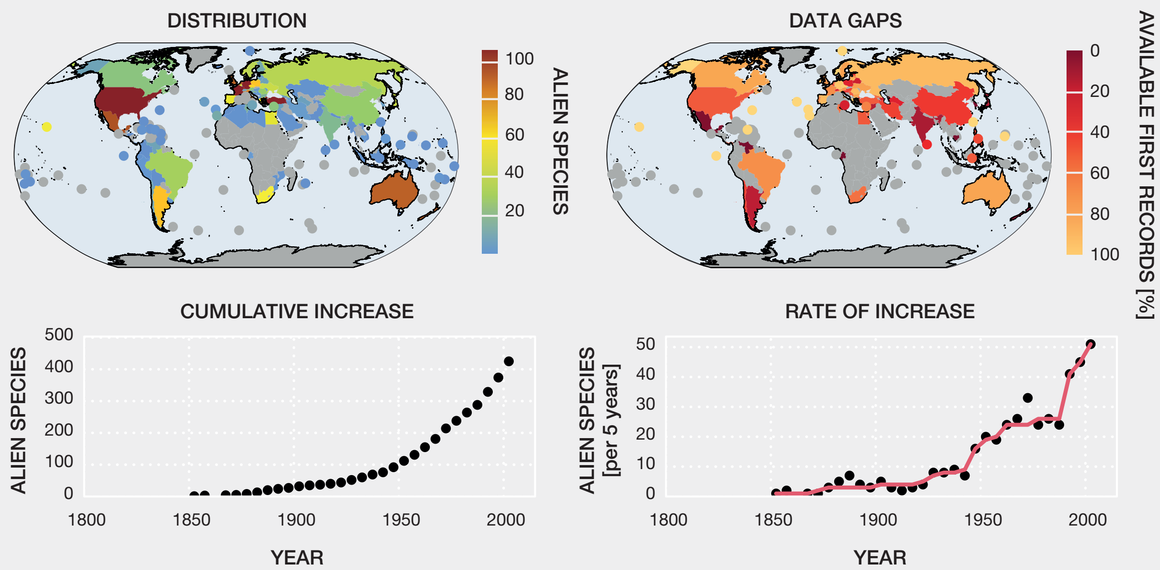

Figure 2 16 **Status, trends and data gaps for other established alien invertebrates.**

Other established alien invertebrates refer to animal groups, which are not covered in the previous sections. The names of the taxonomic groups are listed at the beginning of **section 2.3.1.10**. The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

European seas due to a climate-driven range expansion rather than a human-mediated introduction (Shiganova *et al.*, 2019).

Many earthworm species can be regarded as "ecosystem engineers", that is they play a pronounced role in the creation, modification and maintenance of the upper horizons of the soil habit (Eijsackers, 2011; C. G. Jones *et al.*, 1994; Ponge, 2021). The potential for modifying the soil environment means that earthworms can have a disproportionate impact on the communities that they invade (Hendrix *et al.*, 2008). This is especially true in circumstances where earthworms invade soils that previously had an absent or impoverished earthworm fauna (Frelich *et al.*, 2019). Globally, more than 100 alien earthworm species are documented (Hendrix, 2006) but have mostly been neglected until very recently. For example, earthworm invasions in North America date back to the first European settlers, but because they live underground, they have remained mostly unnoticed (Migge-Kleian *et al.*, 2006). Ongoing invasions of European earthworms into the Upper Midwest of the United States are relatively well documented (Hale *et al.*, 2005) compared to the invasion in the Northeast (Stoscheck *et al.*, 2012; Suárez *et al.*, 2006). Alien earthworms can often be found spreading into habitats where few or no native earthworms exist, such as in North America which has been depauperate in native earthworms since the last glaciation (McCay & Scull, 2019). Similar patterns are believed to exist in the taiga region in Russia and the coniferous forests of Scandinavia (Hendrix, 2006). The earthworm fauna of the North American northeast now includes a few native species (Csuzdi *et al.*, 2017), many alien species from Europe, and a rapidly rising number of species from Asia (Addison, 2009; McCay & Scull, 2019). The tropical earthworm *Pontoscolex corethrurus*, originally native to Guyana, was introduced to tropical and sub-tropical regions worldwide (S. Taheri *et al.*, 2018). *Platydemus manokwari* (New Guinea flatworm) was both unintentionally and deliberately introduced into the soils of many countries and islands, where it leads gregarious attacks on large earthworms and land snails (Sugiura, 2010; Sugiura & Yamaura, 2009). Another flatworm, *Obama nungara* from South America, has been introduced to France (Justine *et al.*, 2020). *Arthurdendyus triangulatus* (New Zealand flatworm) can now be found in Great Britain where it causes declines in native earthworm populations (Murchie & Gordon, 2013).

There is a growing recognition of the influence of alien earthworms in tropical environments as well (Marichal *et al.*, 2012; Ortíz-Ceballos *et al.*, 2019; Potapov *et al.*, 2021; S. Taheri *et al.*, 2018). Earthworm communities in tropical agricultural environments often consist of both native and invasive alien species; however, it is not always clear what role these species are playing, though, without doubt, deforestation, the spread of plantations, landscaping and an expansion of human activity may serve as drivers that facilitate further invasion (Potapov *et al.*, 2021).

Along the south-eastern Pacific coast, there are records for six introduced species of polychaete worms from the families Spionidae and Sabellidae (Fuentes *et al.*, 2020; Moreno *et al.*, 2006). The species *Polydora rickettsi*, *Polydora hoplura* and *Terebrasabella heterouncinata* were accidentally introduced. There is no information regarding the type of introduction for *Boccardia tricuspa*, *Polydora bioccipitalis* and *Dipolydora giardi* (Fuentes *et al.*, 2020). All of them compete with the native species. These introductions also cause negative economic impacts in the aquaculture industry by boring and infesting the shells of cultured molluscs (Fuentes *et al.*, 2020; Moreno *et al.*, 2006; **Chapter 4**, **Box 4.13**).

2.3.1.11 Data and knowledge gaps

Global analyses on invasion trends and status for animals are limited to some taxonomic groups, such as mammals, birds, reptiles, amphibians, fish, land snails, spiders, crustaceans and ants. Many case studies exist on species of other groups, but they provide substantially less information on general patterns.

Data and knowledge gaps are pervasive across all taxonomic groups and geographical levels (**Figure 2.6**; Pyšek *et al.*, 2008; Troudet *et al.*, 2017). Charismatic species such as birds and mammals tend to be more studied while other taxa, such as herpetofauna and invertebrates, have weaker sampling efforts and hence more data gaps (Pyšek *et al.*, 2008; Rocha-Ortega *et al.*, 2021; Troudet *et al.*, 2017). However, even the most intensively studied taxa may not be fully documented at the global scale resulting in geographic biases mainly driven by economic development (Dawson *et al.*, 2017) and linguistic barriers (Angulo *et al.*, 2021; Nuñez & Amano, 2021). The data gaps comprising both taxonomic groups and geographical regions in the marine realm are particularly apparent. Unlike terrestrial and freshwater alien species, marine alien species are mostly unintentionally introduced, and most records are either confined to economically impactful species, or to (relatively) large-sized sessile taxa inhabiting the intertidal or the shallow shelf. Even for these taxa, surveys have not been conducted along region-wide coastlines, leaving most alien taxa undetected and unrecognized. This presents an enormous challenge for understanding the dynamics of these biological invasions and prioritizing conservation and research aims for marine ecosystems (Ojaveer *et al.*, 2015, 2018).

Comprehensive analyses of data and knowledge gaps of alien species occurrences are largely lacking on a global scale. The few global systematic reviews of alien species distributions available for well-studied taxonomic groups such as mammals (Biancolini *et al.*, 2021), birds (E. E. Dyer, Cassey, *et al.*, 2017), reptiles and amphibians (Capinha *et al.*, 2017) indicate large geographic areas of incomplete information. For example, global systematic reviews of studies of first record data for alien amphibians and reptiles (N. J. van Wilgen *et al.*, 2018; **Figures 2.10** and **2.11**) using model-based estimates of the number of alien turtles expected to be introduced but not detected worldwide (García-Díaz *et al.*, 2015), showed consistent spatial gaps. Alien reptiles and amphibians have been understudied in Africa and parts of Asia, whereas the knowledge of alien amphibians and reptiles in Meso- and South America varies by country. These spatial patterns broadly mirror those of native reptiles and amphibians assessed as data-deficient in global International Union for Conservation of Nature (IUCN) Red List of threatened species assessments (Böhm *et al.*, 2013; Stuart *et al.*, 2008) and are very similar for other taxonomic groups.

In some cases, even though large regions are indicated as invaded due to country-level reporting, it is likely that only certain areas of these countries are actually invaded. This coarse scale reporting may cause distorted understanding of global distribution maps of these species by assigning very large territories to invasions while in fact, only smaller areas might be concerned. When numbers of invasive alien species are compiled, large countries are more likely to be tallied as containing species, even if their distributions are not greater than in smaller countries, thus contributing to this bias. Also, species introduced to new parts of a country where they did not previously exist are often not reported as being alien, and therefore, total numbers of alien species are frequently underestimated.

Data documenting invertebrate invasions are grossly incomplete. Earthworms are understudied compared to the impact they have on invaded ecosystems (Hendrix, 2006; Porco *et al.*, 2013). Many invertebrates are small and inconspicuous, and so large numbers of alien invertebrates remain undetected. For example, many Hymenoptera parasitoids have likely invaded regions without being detected likely due to a lack of available expertise and monitoring. The Asian parasitic wasp species *Gryon japonicum* (samurai wasp) was being evaluated for introduction as a biological control agent of *Halyomorpha halys* (brown marmorated stink bug) in North America when researchers discovered that it was already present (Talamas *et al.*, 2015). Addressing this problem not only requires increased survey effort, but also requires increased taxonomic research, since many insect species remain undescribed.

Research efforts are also driven by the actual, perceived, or projected impacts of invasive alien species, with highest-impact species being the most studied (e.g., bivalves, a small number of ants, a few other insects, some crustaceans, most vertebrates), while those causing less conspicuous damage are sometimes neglected (Pyšek *et al.*, 2008). For example, of the 19 highly invasive ant species, only two are extensively studied (over 350 studies each in Web of Science), three are much less covered, and the remaining species are almost entirely ignored (more than 3 per cent of all studies for the 14 other species cumulatively; Bertelsmeier *et al.*, 2016). Such disparities reflect presumed impacts and can potentially bias studies towards species with high expected impacts, but they also reflect the low number of biological invasion researchers and managers relative to the number of insect invasions.

Other factors contributing to data and knowledge gaps include taxonomic uncertainties, inadequate historical records, lack of data mobilization (i.e., making data available and accessible), sharing, and insufficiently applied expertise. Many ecosystems – especially freshwater and marine systems – harbour species that cannot be categorized as either alien or native with any high degree of certainty. In other cases, alien species are wrongly and erroneously assumed to be native and to have a natural cosmopolitan distribution (Carlton, 2009; Jarić *et al.*, 2019). The problem is most severe for small-bodied invertebrates (Marchini & Cardeccia, 2017; Ruiz & Carlton, 2003). Freshwater examples include bryozoans and rotifers, which are ubiquitous in lakes and rivers and have resting stages that are common and abundant in the ballast water of some transoceanic ships (Kipp *et al.*, 2010), but are rarely reported as alien species even in highly invaded aquatic systems (Pociecha *et al.*, 2016; Ricciardi, 2015).

In addition to information on the occurrence of alien populations, the dates of first introduction are unknown for most taxa except for avian and mammalian species (Biancolini *et al.*, 2021; E. E. Dyer, Redding, *et al.*, 2017). In general, more of this temporal information exists for Europe, especially for mammals and birds, while large gaps are found in Central Africa and South Asia. However, in most cases, the proportion of species with available temporal information is far below 50 per cent (Seebens *et al.*, 2020), often including well-studied regions like North America and Europe. Furthermore, there is a severe gap in temporal information for invertebrates all over the world.

More work to address the current knowledge gaps remains to be done. In particular, further genetic research including environmental deoxyribonucleic acid (DNA; Herder *et al.*, 2014; Hunter *et al.*, 2015; Tingley *et al.*, 2019) will contribute to resolving the alien or native status of some species and to uncovering cryptic and unrecognized introductions (Cogălniceanu *et al.*, 2014; Silva-Rocha *et al.*, 2012; Telford *et al.*, 2019).

2.3.2 Plants

This section reports on the temporal trends and status of the distribution of alien and invasive alien plant species for vascular plants (**section 2.3.2.1**), aquatic plants (**section 2.3.2.2**), algae (**section 2.3.2.3**) and bryophytes (**section 2.3.2.4**) as well as data and knowledge gaps (**section 2.3.2.5**).

2.3.2.1 Vascular plants

Trends

The total number of alien plant species established outside of their native ranges worldwide has increased continuously for centuries **(Figure 2.17)**, and first records of alien plants dating back more than one thousand years exist from all over the world (van Kleunen *et al.*, 2019; Wijesundara, 2010). As with many other taxonomic groups, the rate of accumulation for plants rose dramatically in the second half of the nineteenth century, tapering off in the early twentieth century, but increasing steeply after ca. 1970. Indeed, 28 per cent of all established plant records worldwide were recorded for the first time after 1970 **(Figure 2.17)**.

The number of alien plant species introduced is particularly important because plant introductions (whether intentional or unintentional) are a pathway for other invasive alien species introductions such as forest pests and pathogens, microbes, and other hitchhikers (Hulme *et al.*, 2008). The historical flow of alien plant species among continents shows that Europe and temperate Asia are the major donors of established alien plant species to other parts of the world (Drake *et al.*, 1989; van Kleunen *et al.*, 2015). The number of species native to Europe that have been established elsewhere is almost three times higher than expected (van Kleunen *et al.*, 2015). North America is also over-represented, with 57 per cent more species donated than expected based on native continental richness. In contrast, the continents in the Southern Hemisphere are all under-represented as donors of alien species. This suggests that, at least for plants, the "Old World *versus* New World" dichotomy (a classical concept in biological invasions suggesting that "Old World" biota were more likely to invade other parts of the globe due to traits they developed in close association with humans in their native ranges; Di Castri, 1989) needs to be replaced by a Northern Hemisphere *versus* Southern Hemisphere dichotomy for the donor continents of established alien plants (van Kleunen *et al.*, 2015).

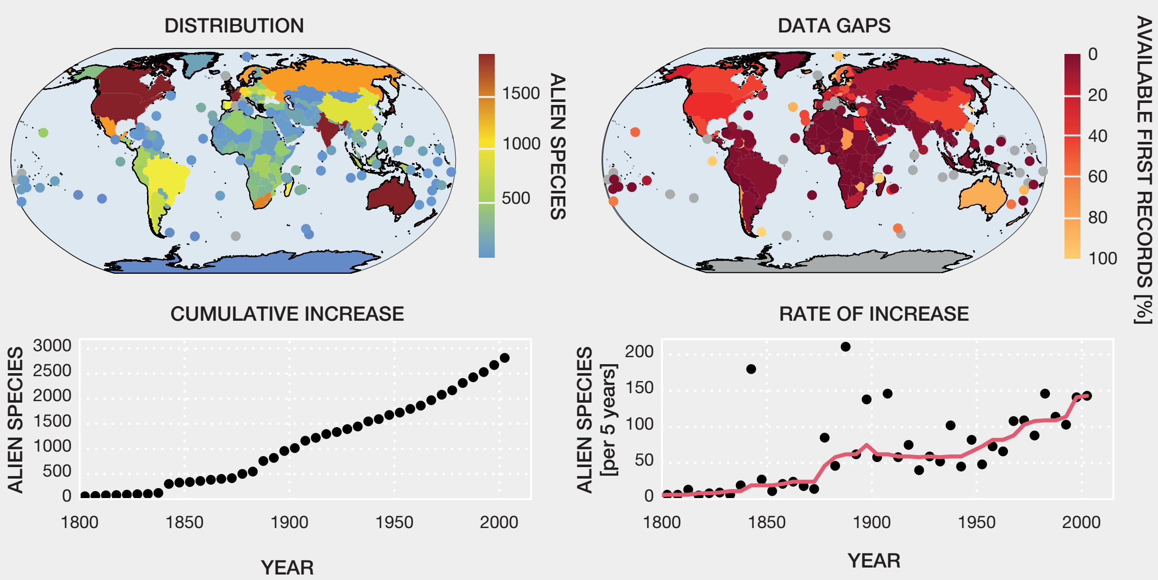

Figure 2 17 **Status, trends, and data gaps for established alien vascular plants.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

While North America has accumulated the greatest number of established alien species, the Pacific islands show the fastest increase in species numbers with respect to land area suggesting that Pacific islands have the highest vulnerability to invasions of all areas globally. Oceanic islands harbour more established alien plant species than similarly sized mainland regions, a phenomenon traditionally attributed to the niche space being unsaturated by native species or to a greater frequency of introductions (Moser *et al.*, 2018; van Kleunen *et al.*, 2015). Given the high concentration of endemic species on most oceanic islands, the large numbers of established alien species constitute a serious threat to global biodiversity (Fernández-Llamazares *et al.*, 2021; Pyšek, Blackburn, *et al.*, 2017; van Kleunen *et al.*, 2015).

Status

Currently, the total number of established alien plant species (13,939 species; van Kleunen *et al.*, 2019) indicates that at least 4 per cent of all known vascular plant species (337,137 species; The Plant List, 2015) have become established outside their natural ranges because of human activity. In total, 12,345 established alien species are reported from mainland regions globally and 8,019 from islands (Pyšek, Pergl, *et al.*, 2017).

The cool temperate forest and woodland regions have the highest richness of established alien plant species (6,586 species), followed by tropical (equatorial 4,690 species, and savanna 4,843 species), and warm temperate regions (4,649 species). In total, temperate regions harbour 9,036 established alien species relative to 6,774 for tropical zones, 3,280 in the Mediterranean regions, 3,057 in subtropical regions, and 321 in Arctic regions. When the total number of established alien species is standardized to the area of each region by comparing species accumulation rates with area, it appears that colder temperate and Mediterranean regions are more heavily colonized by alien species while more arid regions have fewer (**Figure 2.17**; Pyšek, Pergl, *et al.*, 2017).

Hotspots of relative alien species richness (i.e., the per cent of established alien species in the total regional flora) appear on both the western and eastern coasts of North America, north-western Europe, South Africa, south-eastern Australia, New Zealand, and India. South Africa, India, California (United States), Cuba, Florida (United States), Queensland (Australia) and Japan have the highest absolute values of established alien species (Essl *et al.*, 2019; Pyšek, Pergl, *et al.*, 2017). The mainland regions with the highest numbers of established alien species include several Australian states (New South Wales is highest in established alien richness on this continent) and several North American regions such as California, which has 1,753 established alien plant species. High levels of island colonization by established alien plants are concentrated in the Pacific region, but also occur on individual islands across all oceans. About one quarter (26 per cent) of the islands investigated by Essl *et al.* (2019) now have more established alien species than native species. England, Japan, New Zealand, and the Hawaiian archipelago harbour most established alien plants among islands or island groups (Pyšek, Pergl, *et al.*, 2017). Numbers of established alien species are closely correlated with those of native species and also with those of invasive alien species. There is also a faster increase in the numbers of established alien species with area on islands than in mainland regions, indicating a greater vulnerability of islands to alien species establishment (Essl *et al.*, 2019; Pyšek, Pergl, *et al.*, 2017).

Among vascular plants, the introduction of alien ferns is certainly less investigated and only one global assessment for alien ferns exists (E. J. Jones *et al.*, 2019). This study lists 157 alien ferns which are found in all climatic zones except the Arctic and Antarctic and on all continents. High numbers of alien ferns were reported for New Zealand, Hawaii, India and Europe.

In terms of plant families, rankings by absolute numbers of established alien species reveal that Asteraceae (1,343 species), Poaceae (1,267) and Fabaceae (1,189) contribute most to the global established alien flora. Comparing the number of established alien species in a family to its total global richness reveals that some of the large species-rich families are over-represented among established alien species (e.g., Poaceae, Fabaceae, Rosaceae, Amaranthaceae, Pinaceae), some under-represented (e.g., Euphorbiaceae, Rubiaceae), whereas Asteraceae, which has the highest richness of established alien species, reaches an expected value based on its global species richness. A significant phylogenetic signal indicates that some plant families have a higher potential for species to establish (Pyšek, Pergl, *et al.*, 2017). *Solanum* (112 species), *Euphorbia* (108) and *Carex* (106) are the richest genera in terms of established alien species. Some families are disproportionately over-represented by alien species on islands (i.e., Arecaceae, Araceae, Acanthaceae, Amaryllidaceae, Asparagaceae, Convolvulaceae, Rubiaceae, Malvaceae), but significantly fewer families are over-represented on mainlands (e.g., Brassicaceae, Caryophyllaceae, Boraginaceae). On islands, the genera *Cotoneaster*, *Juncus*, *Eucalyptus*, *Salix*, *Hypericum*, *Geranium*, and *Persicaria* are over-represented, while on the mainland *Atriplex*, *Opuntia* (pricklypear), *Oenothera*, *Artemisia*, *Vicia*, *Galium*, and *Rosa* are relatively richer in established alien species (Pyšek, Pergl, *et al.*, 2017).

The 10 most widely distributed established alien plants globally occur in at least 35 per cent of the world's regions. Other species such as *Sonchus oleraceus* (common sowthistle) occur in 48 per cent of the regions corresponding to 42 per cent of the globe. Additional widely distributed established alien species are *Oxalis corniculata* (creeping woodsorrel), *Portulaca oleracea* (purslane), *Eleusine indica* (goose grass), *Chenopodium album* (fat hen), *Capsella bursa-pastoris* (shepherd's purse), *Stellaria media* (common chickweed), *Bidens pilosa* (blackjack), *Datura stramonium* (jimsonweed), and *Echinochloa crus-galli* (barnyard grass). However, the ranking for invasive alien species differs among global databases because the data differ depending on the source used. The GloNAF database highlights *Lantana camara* (lantana,120/349 regions for which data on invasive status are known), *Calotropis procera* (apple of sodom, 118), *Pontederia crassipes* (water hyacinth, 113), *Sonchus oleraceus* (108) and *Leucaena leucocephala* (leucaena, 103) as the most distributed invasive alien species (Pyšek, Pergl, *et al.*, 2017), while GRIIS (Pagad *et al.*, 2022) provides a different ranking **(Table 2.15)**.

Table 2 15 **Top 10 most widespread invasive alien vascular plant species worldwide.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien vascular plant species rather than their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Pontederia crassipes* (water hyacinth)**	74	*Robinia pseudoacacia* (black locust)	45
***Lantana camara* (lantana)**	69	*Chromolaena odorata* (Siam weed)	43
***Leucaena leucocephala* (leucaena)**	55	*Pistia stratiotes* (water lettuce)	41
***Ricinus communis* (castor bean)**	47	*Erigeron canadensis* (Canadian fleabane)	38
***Ailanthus altissima* (tree-of-heaven)**	46	*Cyperus rotundus* (purple nutsedge)	37

Box 2 2 **Cacti, grasses and woody species: A global assessment of trends and status of alien and invasive alien species.**

Cacti (Cactaceae, about 1,922 species), grasses (Poaceae, about 11,000 species) and woody species are among the most studied species from a plant invasion perspective.

Cacti, native to the Americas, were among the first plants brought back by European explorers from the Americas in the fifteenth century. Most cacti (about 1,600 species, 81 per cent of the family) have been introduced outside their native ranges *via* the horticultural trade, especially recently due to higher volumes of e-commerce (**Glossary**; Novoa *et al.*, 2017), rapidly increasing the number of established alien cactus species **(Figure 2.18)**. However, only 3 per cent of species in Cactaceae (57 species) are currently considered as invasive alien species (Novoa *et al.*, 2015), with *Opuntia ficus-indica* (prickly pear) being the most widespread **(Figure 2.19)**. Although countries such as France, India or the United States support many established alien cacti **(Figure 2.20)**, there are three main hotspots for invasive alien cacti globally: South Africa (35 species recorded), Australia (26 species) and Spain (24 species). Most invasive alien cacti are native to Argentina, Mexico, and North America, which are roughly bioclimatically similar to the invaded regions. Other large regions, such as China, North- and South-East Asia, and Central Africa that are not intensively invaded by cacti have suitable climates for invasive cacti and therefore are at risk of future invasions (**Glossary**; Novoa *et al.*, 2015).

Grasses have been introduced outside their native ranges for horticulture, soil stabilization, as food and fodder, as biofuel, or as raw materials. Most remarkably, forage grasses have been a major focus of plant introduction programmes across large areas (Visser *et al.*, 2016). Perhaps as a result of such large introduction events, the number of established alien grass species has been intermittently increasing since the nineteenth century **(Figure 2.18)**. Currently, 1,226 alien grass species are reported as established globally (Pyšek, Pergl, *et al.*, 2017).

Box 2 2

Regions with the highest numbers of established alien grasses are Indonesia, Hawaii, Madagascar, New Zealand, tropical Africa, tropical South America and the southern United States **(Figure 2.20)**. Among all grasses, tall-statured grasses (defined as grass species that maintain a self-supporting height taller than or equal to 2 meters; 929 species) are 2–4 times more likely to establish than shorter grasses (Canavan *et al.*, 2019). This is due in part to their rapid growth rates and capacity to accumulate biomass. Tropical Africa (especially islands in the Western Indian Ocean) is the main hotspot of established alien tall statured grasses, with this group accounting for 30 to 70 per cent of all established alien grasses. The Caribbean is another such hotspot (Canavan *et al.*, 2019). Overall, 80.6 per cent of all tall statured grasses are woody bamboos, of which *Bambusa vulgaris* (common bamboo) is the most widespread species **(Figure 2.20)**.

Many **woody species** (shrubs and trees) are among the most widespread and damaging invasive plants (D. M. Richardson & Rejmánek, 2011). While there is no precise data available on the number of established woody species, D.M. Richardson and Rejmánek (2013; 2011) compiled a global database of 751 invasive alien woody species, comprised of 434 trees and 317 shrubs in 90 plant families and 286 genera. These alien species were introduced outside of their native ranges through many pathways including horticulture (62 per cent of invasive woody species: 196 trees and 187 shrubs), forestry (13 per cent), food (10 per cent), and agroforestry (7 per cent). Regions with the largest numbers of woody invasive alien species are North America (212), Pacific Islands (208), Australia (203), Southern Africa (178), Europe (134), and Indian Ocean Islands (126). Taxa within the genera *Acacia* and *Pinus* (Pine) comprise a large portion of the woody invasive alien species globally. In particular, *Pinus* (comprising 111 tree and shrub species, only one of which has its natural range confined to the Northern Hemisphere) have been widely introduced and planted in many areas well outside their native range and are among the most widely used forestry species worldwide (D. M. Richardson *et al.*, 1994). At least 30 *Pinus* species are known to be established alien species and 21 invasive alien species (D. M. Richardson, 2006). *Pinus contorta* (lodgepole pine) is one of the most invasive plantation trees **(Figure 2.19)**. Native to northwest North America, it is established in Great Britain, Ireland, and Russia, and is an invasive alien species in Argentina, Australia, Chile, New Zealand, and Sweden (Langdon *et al.*, 2010). *Pinus* invasions were first recorded in South Africa in 1855, in New Zealand in 1880 and in Australia in the 1950s (20-30 years after the first large plantations were established), and most research on *Pinus* invasions has been done in those countries (Simberloff *et al.*, 2009). However, because of a recent increase in commercial *Pinus* plantations in South America (Argentina, Brazil, Chile, and Uruguay are the countries having the greatest area of planted *Pinus*), *Pinus* invasions are currently an emerging problem on the continent and are predicted to increase rapidly in the next few decades (D. M. Richardson *et al.*, 2008). *Acacias* (about 1,350 species), especially Australian acacias (species within the genus *Acacia* that are native to Australia, about 1,012 species), have also been widely introduced outside their native ranges for centuries (D. M. Richardson *et al.*, 2011). At least 386 Australian acacias have been introduced outside Australia, of which 71 are recorded as established alien species and 23 as invasive alien species. The extent of Australian acacia invasions is likely to increase in the future, given that climatic models have suggested that a third of the world's terrestrial surface is climatically suitable. For example, *Acacia dealbata* (acacia bernier; **Figure 2.19**) is currently recorded as an invasive alien species in seven countries (D. M. Richardson & Rejmánek, 2011). Since it has been introduced widely outside of Australia, further accounts of its invasion are likely.

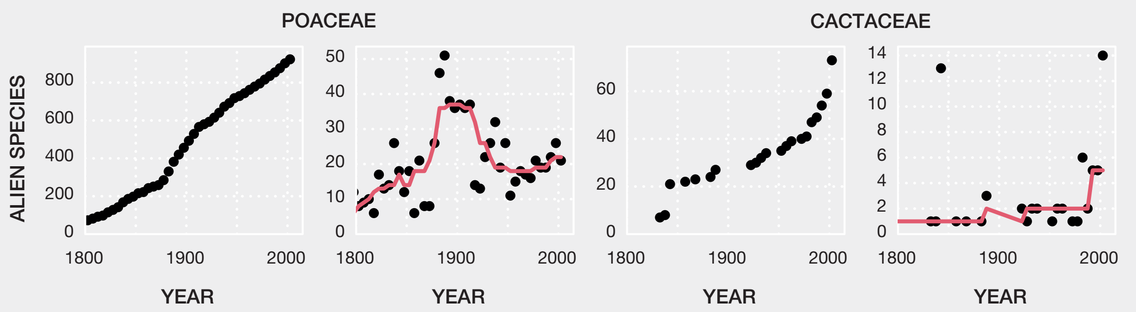

Figure 2 18 **Trends in numbers of established alien species for Poaceae and Cactaceae.**

Cumulative numbers (left panels) and number of established alien species per five-year intervals (right panels). Numbers shown underestimate the true extent of alien species occurrences due to a lack of data. Smoothed trends (line) are calculated as running medians (**section 2.1.4** for further details about data sources and data processing). A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Box 2 2

Figure 2 19 **Examples of the most widespread invasive cacti, grasses and woody species.**

Opuntia ficus-indica (prickly pear; top left) is the most commercially important cactus and is recorded as invasive in 26 countries worldwide. *Bambusa vulgaris* (common bamboo; top right) is the most widely cultivated bamboo and recorded as invasive in 5 countries. *Pinus contorta* (lodgepole pine; bottom left) is one of the most invasive plantation trees and it is recorded as invasive in 5 countries. *Acacia dealbata* (acacia bernier; bottom right) was introduced to many regions for multiple purposes and is now a widespread invasive alien species in 7 countries. Photo credit: Nicole Pankalla, Pixabay – under license CC BY 4.0 (top left) / Bishnu Sarangi, Pixabay – CC BY 4.0 (top right) / Walter Siegmund – CC BY 4.0 (bottom left) / Ulrike Leone, Pixabay – CC BY 4.0 (bottom right).

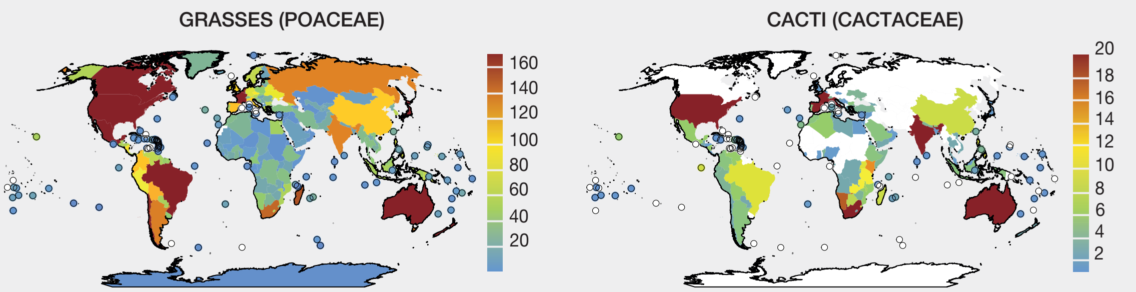

Figure 2 20 **Numbers of established alien grasses and cacti worldwide.**

Colours indicate established alien species of the families Poaceae and Cactaceae per region, including terrestrial, freshwater and marine species. For islands, numbers are shown as dots for visualization. White areas on land denote that information is lacking. Note that the legend scale varies among panels (**section 2.1.4** for further details about data sources and data processing). A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

2.3.2.2 Aquatic plants

Trends

The first records of alien aquatic plants date back to the eighteenth century, becoming more numerous by the early 1900s (Brundu, 2015b; Chomchalow, 2011; Gettys, 2019; M. P. Hill *et al.*, 2020; Hussner *et al.*, 2010). As modelled by Seebens, Bacher, *et al.* (2021), the rate of first records for alien aquatic plants increased post-1950, especially after 1980 when the ornamental plant trade increased (Hrivnák *et al.*, 2019; Hussner *et al.*, 2010; Nunes *et al.*, 2015), and again after 2008 when aquatic detection improved with the development of environmental DNA technology. Both the numbers and rates of established alien aquatic plants are projected to continue to increase until 2050 (Seebens, Bacher, *et al.*, 2021).

Status

Of the 13,168 established alien plant species reported in the GloNAF database, less than 1 per cent are aquatic (Pyšek, Pergl, *et al.*, 2017). However, comprehensive assessments of aquatic alien plants globally are lacking. Still, some aquatic plant species are prominent invasive alien species. Originally from the tropical zone of South America, *Pontederia crassipes* (water hyacinth), is one of the world's most prevalent invasive alien aquatic plants. This free-floating vascular plant has invaded freshwater systems in 62 countries, from 40°N to 40°S (Pan *et al.*, 2011) and, according to recent climate change models, its distribution may expand into higher latitudes as temperatures rise. It is prevalent in tropical and subtropical waterbodies where nutrient concentrations are often high due to agricultural runoff, deforestation, and insufficient wastewater treatment. There are no records of *Pontederia crassipes* first introductions, but many populations are well established and persistent despite control efforts (Coetzee *et al.*, 2017; Villamagna & Murphy, 2010). Sheppard *et al.* (2006) provide an evaluation of several aquatic invasive alien plant species distributions and status in Europe. For example, *Azolla filiculoides* (water fern), a small annual floating fern (hydrophyte), became established in slow moving and still water in ponds, canals, dikes and lakes, following escape from aquaria and botanical gardens in the mid-nineteenth century. The plant is now widespread in Central and Western Europe, South Africa, China and Australasia. Species from the Americas such as *Ludwigia grandiflora* (water primrose), *Ludwigia peploides* (water primrose), and aquatic perennial herbs (hydro-hemicryptophytes) are classified as invasive alien species in Europe. *Crassula helmsii* (Australian swamp stonecrop), originally from Australia and New Zealand, arrived in the United Kingdom in the 1950s and is known as an invasive alien species in the United Kingdom and the Kingdom of the Netherlands. *Elodea canadensis* (Canadian pondweed) and *Elodea nuttallii* (Nuttall's waterweed), both native to North America, are the most widespread alien aquatic plants in Europe. Introduced in the mid-1800s, *Elodea canadensis* spread along river systems throughout Europe in the latter half of the century and now occurs in many other countries worldwide. In the early twentieth century, *Elodea canadensis* was replaced by *Elodea nuttallii* in many regions. *Elodea nuttallii* may in turn begin to be replaced by another invasive alien hydrocharitacean species, *Lagarosiphon major* (African elodea), in the United Kingdom (Brundu, 2015a). *Myriophyllum aquaticum* (parrot's feather), from tropical and subtropical South America, is the dominant invasive alien aquatic plant in Europe. First introduced into France (1880) and then Portugal (1935) as an aquarium escapee, *Myriophyllum aquaticum* is also present in the United Kingdom and the Kingdom of the Netherlands and is probably more widespread as it was sold as an "oxygenating plant" until 2016. It is also a major weed in the United States, Australasia, Southern Africa, and Asia.

Among marine vascular plants, the seagrass *Zostera japonica* (dwarf eelgrass) was introduced to the Pacific Northwest in the mid-1900s likely *via* oyster aquaculture and has since spread and negatively impacted native *Zostera marina* (eelgrass) and ecosystem processes (Shafer *et al.*, 2014). Additionally, *Halophila stipulacea* (halophila seagrass) was introduced to the Mediterranean Sea through the Suez Canal where it is now widespread (Willette *et al.*, 2014). More recently, *Halophila stipulacea* was introduced to the Caribbean Sea where it is spreading and is described as the world's first globally invasive marine angiosperm (Willette *et al.*, 2014; Winters *et al.*, 2020).

2.3.2.3 Algae

In this section, algae are comprised of taxa of the phyla Rhodophyta, Chlorophyta, Charophyta, Cryptophyta, Euglenozoa, Haptophyta, Foraminifera, Ciliophora, Ochrophyta, Myzozoa and Cercozoa. Other groups of microorganisms are covered in **section 2.3.3**.

Trends

Globally, many alien green, brown, and red marine algae have been reported, with steep increases **(Figure 2.21)** in reports of large macroalgae invaders since the mid-twentieth century (Carlton & Eldredge, 2009; Fuentes *et al.*, 2020; Ribera & Boudouresque, 1995; J. E. Smith, 2011; Vaz-Pinto *et al.*, 2014; Villaseñor-Parada *et al.*, 2018; S. L. Williams & Smith, 2007). The high rate of increase since this time likely reflects increased global shipping after the invention of containerized transport in 1956. A study on the global distribution of 97 marine algae with known invasion histories revealed that hotspots of future occurrences are in East Asian and European waters, largely reflecting high shipping intensities of enclosed seas (Seebens *et al.*, 2016).

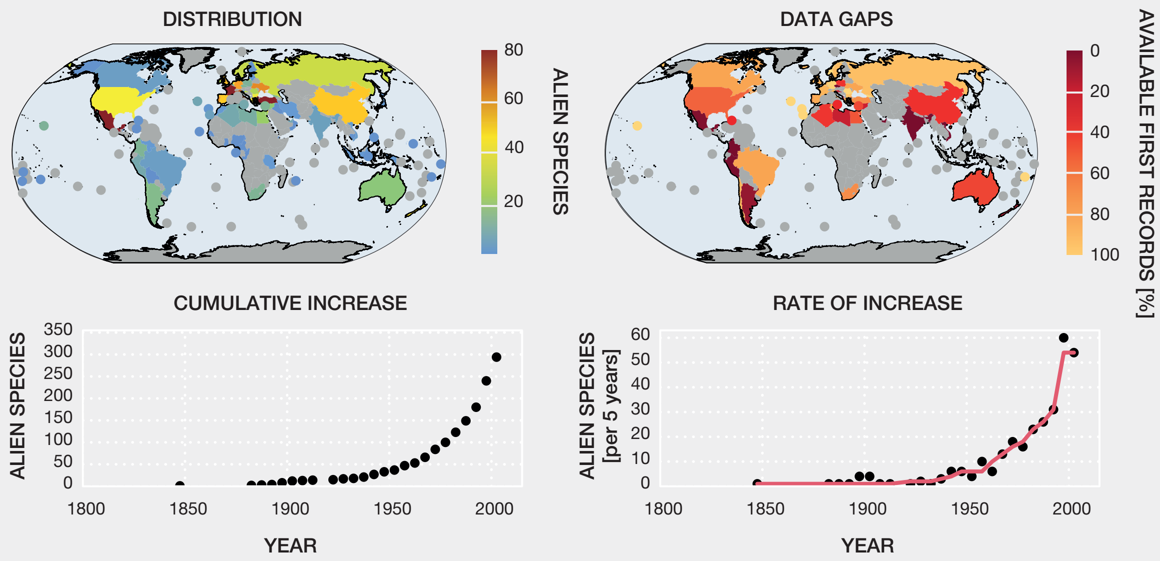

Figure 2 21 **Status, trends, and data gaps for established alien algae.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

The unresolved tensions between using alien species for aquaculture and their potential ecological impacts are well-represented in the history of seaweed invasions. In the 1970s, a suite of alien seaweeds was introduced to the Hawaiian Islands for mariculture, including *Kappaphycus striatus* (Indo-Pacific red algae) and *Gracilaria salicornia* (red alga), and the tropical Atlantic *Hypnea musciformis* (hypnea). In subsequent decades, these algae spread across the Hawaiian Islands. *Kappaphycus* (red alga) is reported to achieve over 50 per cent cover on some Hawaiian coral reefs. Efforts to remove alien seaweeds from Hawaiian reefs are ongoing.

Status

Examples of significant algal invasions with well-documented ecological and economic impacts include a variety of alien species native to Asia, such as *Sargassum muticum* (wire weed), *Codium fragile* (dead man's fingers), *Grateloupia turuturu* (devil's tongue weed), *Gracilaria vermiculophylla* (black wart weed), and *Asparagopsis armata* (Harpoon weed) – all now found on many continental margins around the world. Less widely distributed but even more notorious is *Caulerpa taxifolia* (killer algae), toxic to certain herbivores. More broadly distributed alien macroalgae are not necessarily more likely to succeed in new regions than more narrowly distributed species (S. L. Williams & Smith, 2007). For example, the genus *Capreolia* (red algae), considered endemic to Australasia, has been found on the coast of central Chile, based on molecular and morphological analysis (Boo *et al.*, 2014). *Pyropia koreana* (red algae) described previously from Korea, has been reported in the Mediterranean Sea (Vergés *et al.*, 2013) and New Zealand (Nelson *et al.*, 2014) and was detected using molecular analysis. Finally, *Chondracanthus chamissoi* (yuyo), considered endemic to the south-central coast of Chile, has been reported, through molecular analysis, in France, Japan, and Korea, where it shows important morphological variations (M. Y. Yang *et al.*, 2015; **Table 2.16**).

The cultivation of algae has facilitated the transfer of native species within country borders but still outside its historical range of distribution. For example, the macroalga *Gracilaria chilensis* (red seaweed), native to the south-central coast of Chile, has been extensively cultivated more than 640 km from its northern limit of distribution (Guillemin *et al.*, 2008; Santelices, 1989), resulting in established alien populations from the escape of vegetative propagules from aquaculture facilities (Castilla & Neill, 2009; Guillemin *et al.*, 2008; Villaseñor-Parada & Neill, 2011). Moreover, alien mollusc aquaculture has been identified as an introduction vector for many invasive macroalgae (Ribera Siguan, 2003; S. L. Williams & Smith, 2007). Indirect evidence suggests that

Table 2 16 **Top 10 most widespread invasive alien algae species worldwide.**

The number of regions where the species is recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien algae species rather than their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
Undaria pinnatifida **(Asian kelp)**	9	*Gracilaria vermiculophylla* (black wart weed)	5
Sargassum muticum **(wire weed)**	8	*Coscinodiscus wailesii* (diatom)	5
Caulerpa taxifolia **(killer algae)**	7	*Dasysiphonia japonica* (siphoned Japan weed)	5
Caulerpa cylindracea **(green algae)**	6	*Alexandrium tamarense* (dinoflagellate)	4
Codium fragile **(dead man's fingers)**	6	*Alexandrium minutum* (dinoflagellate)	4

several species of alien macroalgae have been introduced by aquaculture of *Magallana gigas* (Pacific oyster) in Europe (Krueger-Hadfield *et al.*, 2017; Lang & Buschbaum, 2010; Mineur *et al.*, 2007), North America (Mathieson *et al.*, 2003) and South America (D. E. Bustamante & Ramírez, 2009; Croce & Parodi, 2014). Filamentous alien species such as *Polysiphonia morrowii*, or alien species with filamentous stages in their life cycle, such as the "*Falkenbergia* phase" of *Asparagopsis armata* (Harpoon weed) or the "*Vaucheroid* phase" of *Codium fragile* (dead man's fingers), benefit from the rugosities in the shell of *Magallana gigas* where they can pass unobserved.

Alien macroalgae species themselves can serve as an introduction vector for other alien species that live as epiphytes in the thallus. For example, in many ecosystems where *Codium fragile* (dead man's fingers) has been introduced, its most conspicuous epiphyte is the Asian macroalgae *Melanothamnus harveyi* (Harvey's siphon weed; e.g., González & Santelices, 2004; E. Jones & Thornber, 2010; Schmidt & Scheibling, 2006; Villaseñor-Parada & Neill, 2011). Apparently, *Melanothamnus harveyi* is a secondary introduction associated with *Codium fragile*. Native species may also play an important role in the spread of alien species. For example, *Schottera nicaeensis* (red algae) and *Asparagopsis armata* (Harpoon weed) are invasive alien species in the Pacific southeast coast, and they have been found as epiphytes in drifting thalluses of the buoyant macroalgae *Durvillaea antarctica* (cochayuyo), becoming a potential dispersal mechanism for these species (Macaya *et al.*, 2016). For example, the release of reproductive fragments adrift has been identified as alternative dispersal strategies in *Codium fragile* (Villaseñor-Parada *et al.*, 2013) and *Mastocarpus latissimus* (Oróstica *et al.*, 2012).

2.3.2.4 Bryophytes

Trends

Cumulative numbers of first records grew slowly until 1950 and have since increased rapidly worldwide **(Figure 2.22)**, particularly in Oceania and Europe (Essl *et al.*, 2013).

Status

The most comprehensive assessment of alien bryophytes compiled data from 82 locations on five continents in both hemispheres (Essl *et al.*, 2013). To date, 139 species of bryophytes are considered alien in at least one of the regions studied, of which 79 are established, 19 are casual and 41 are cryptogenic (of uncertain origin; **Glossary**) occurrences. Of these, 106 are mosses, 28 liverworts, and 5 hornworts. Only 18 species (i.e., 13 per cent) are recorded as alien from at least five regions, with the most widespread being *Campylopus introflexus* (heath star moss; the best documented invasion, introduced to the United Kingdom in 1941 and coastal Europe in 1954 and currently extending to Russia in the east and the Mediterranean in the south), *Kindbergia praelonga* (common feather moss), *Lunularia cruciata* (crescent-cup liverwort), *Orthodontium lineare* (cape thread-moss), and *Pseudoscleropodium purum* (neat-feather moss). The two most important pathways for bryophyte introductions are unintentional imports as hitchhikers on ships and planes and as epiphytes on ornamental plants and other horticultural supplies with 34 and 27 species, respectively. Most alien bryophytes occur in human-made habitats, such as ruderal sites, roadsides, and lawns, while only a few natural ecosystems such as forests and rocky outcrops regularly harbour alien bryophytes (Essl *et al.*, 2013).

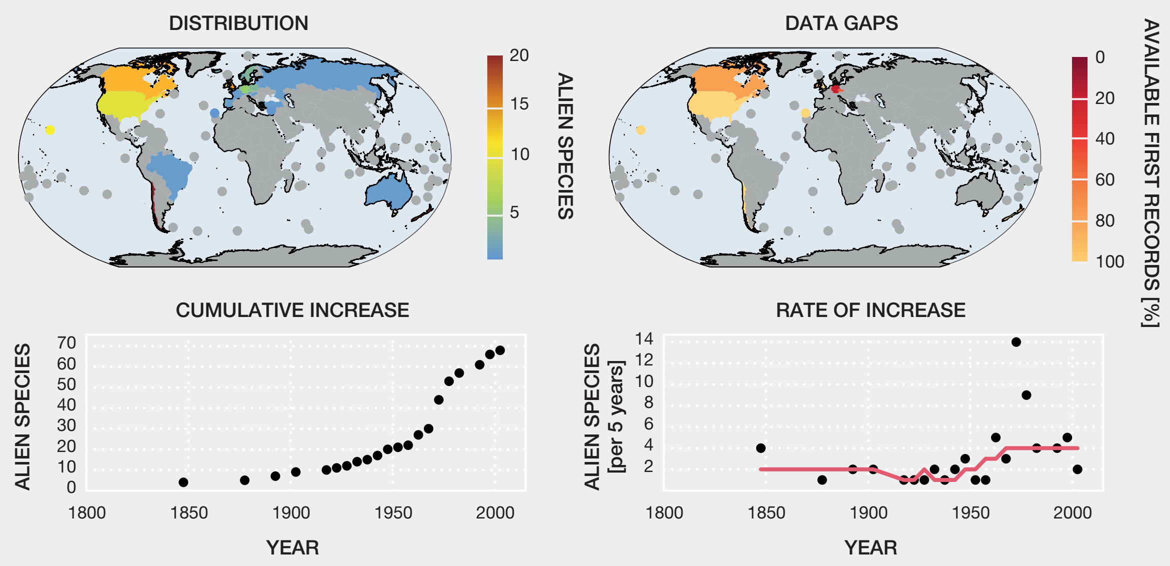

Figure 2 22 **Status, trends, and data gaps for established alien bryophytes.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend line is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Among locations of the Northern Hemisphere, the highest numbers of alien bryophytes are recorded for the Hawaiian Islands, United States and United Kingdom (22 species), followed by British Columbia, Canada (13 species), Ireland (11 species), California, United States (10 species) and France (10 species). In the Southern Hemisphere, most alien bryophyte species are recorded on islands (South and North Islands of New Zealand, 27 species each; St. Helena, 22 species). Continental South America, Asia and Africa have much lower numbers of alien bryophytes, from three to six species (Essl *et al.*, 2013). In general, islands are more invaded by alien (and cryptogenic) bryophytes than continental regions (Essl *et al.*, 2013). For invasive alien bryophytes, GRIIS (Pagad *et al.*, 2022) lists only two species that occur in more than one region, *Campylopus introflexus* (heath star moss) and *Orthodontium lineare* (cape thread-moss), each occurring in two regions.

2.3.2.5 Data and knowledge gaps

The GloNAF database and associated analyses (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015, 2019) make it possible to quantify the proportion of a continental area for which data on established alien vascular plants are available (e.g., **Box 2.2**). GloNAF 1.1 covers more than 83 per cent of the world's ice-free terrestrial surface in terms of regions (n = 843) for which alien floras are available, but there is great variation in the geographic coverage among the continents defined by the Biodiversity Information Standards (TDWG, 2021). There is nearly complete data coverage, in terms of the proportion of individual regions having data on their alien floras, for Australasia (99.5 per cent of regions at the country, state, district or island level have information on alien flora), Africa (98.6 per cent), North America (95.9 per cent), South America (95.8 per cent) and Antarctica (90.2 per cent). The continents with lower coverage are tropical Asia (68.5 per cent), and particularly temperate Asia (54.8 per cent), where data are missing primarily for parts of Russia. The lack of data on alien floras for some regions of the European part of Russia also results in rather low coverage for Europe as a whole (63.8 per cent of the continent area). Data on alien plants are available for about half of the total area of the Pacific islands (49.1 per cent). However, good geographical coverage does not mean the information on the alien plants for a given region is complete; there can be data gaps even for well-studied regions (Pyšek *et al.*, 2008), as well uncertainties about a species status. Notably, identification of alien species is challenging for taxa with a distribution over more than one continent, for which no global identification key is available, and especially when the origin of the alien plant is unknown, such as for Cyperaceae, *Hydrocotyle* or *Myriophyllum*. The quality and completeness of individual datasets also vary greatly, as does the assessment of the status of alien species, habitat affiliations, first records and pathways **(Figure 2.22)**.

Ideally, records of alien plants occurrences would be collected following broadly accepted standards that reflect the research infrastructure and resources (Latombe *et al.*, 2017; **Chapter 6**, **section 6.6.2.3**).

Similarly, comprehensive databases such as the GloNAF database are not available for bryophytes or algae, severely limiting the potential for a thorough assessment of the trends and status for these groups. While alien bryophytes in Central and Western Europe and North America are well-documented, data on alien bryophytes on all other continents, and particularly in the tropics, are rarely available (Essl *et al.*, 2013). The number of algal invasions worldwide is poorly known due to low research efforts. In addition, comparatively high taxonomic uncertainty makes it difficult to compare species identities among studies. Many hundreds of seaweed species bear the same name around the world but are regarded as naturally distributed. These species doubtless represent a mixture of species complexes peppered with many overlooked invasions. Furthermore, the original native ranges are often unknown, making it impossible to determine whether populations are native or alien in that region. As a consequence, many populations of algae and bryophytes species can only be classified as cryptogenic and a comprehensive assessment of the current status of their alien distributions remains elusive.

Finally, the aforementioned databases provide regional lists of alien taxa without information on their precise spatial distributions. In large countries it is especially common that a reported species occurs in only part of the country. Occurrence datasets like the GBIF hold such spatially explicit data but to date report only incomplete information on the biogeographic status of taxa, that is, whether a species is native or alien (C. Meyer *et al.*, 2016). Additionally, like all global databases, GBIF records for plants are biased in terms of taxonomy, space, and time (A. C. Hughes *et al.*, 2021; C. Meyer *et al.*, 2016; Troudet *et al.*, 2017). However, new methods are emerging that allow the use of probabilistic tools to estimate the biogeographic status of occurrence records (Arlé *et al.*, 2021).

2.3.3 Fungi and microorganisms

This section reports on the temporal trends and status of the distribution of alien and invasive alien species for fungi (**section 2.3.3.1**) and the group of Chromista, bacteria and viruses (**section 2.3.3.2**) as well as data and knowledge gaps (**section 2.3.3.3**). In this chapter the group of microorganisms is split into "fungi" (**section 2.3.3.1**) with the phyla Ascomycota, Chytridiomycota, Basidiomycota, Microsporidia, and Zygomycota, and "Chromista, bacteria and viruses" (**section 2.3.3.2**) with the taxonomic groups Oomycota, Actinobacteria, Chlamydiae, Cyanobacteria, Firmicutes, Proteobacteria, and viruses. Other groups of microorganisms are covered in **section 2.3.2.3**. Note that there can be a high degree of uncertainty about to the status of microorganisms as native or alien.

2.3.3.1 Fungi

Trends

Fungi comprise an immensely diverse biological kingdom that forms complex interactions at multiple ecological levels. Fungal invasions are increasingly recognized as key drivers of wildlife mortality and population declines for amphibians, bats, bees, soft coral, and other organisms (Fisher *et al.*, 2012). Introduction of undesirable alien fungi such as those producing repellent smells or toxic compounds, is also problematic (Parent *et al.*, 2000; A. Pringle & Vellinga, 2006). Negative impacts of plant diseases caused by fungal invasions have resulted in widespread ecosystem disruptions that indirectly impact the function of forests, streams, and other natural environments (Anderson *et al.*, 2004; Scott *et al.*, 2019; **Chapter 4**, **section 4.3.1**) such as *Hymenoscyphus fraxineus* (ash dieback; **Table 2.17**) causing ash dieback in Europe. In addition, alien fungal pathogens have severe negative impacts on agricultural crops (**Chapter 4**, **section 4.4.1**). Examples include *Phytophthora ramorum* (sudden oak death; Thakur *et al.*, 2019), *Phyllosticta citricarpa* (citrus black spot; Guarnaccia *et al.*, 2019), *Phakopsora pachyrhizi* (soybean rust; Dean *et al.*, 2012) or *Pyricularia oryzae* (rice blast disease; Fones *et al.*, 2020).

With an increasingly connected world, the rate at which alien fungi are recorded is accelerating (Bebber *et al.*, 2013; Desprez-Loustau, 2009; Fisher *et al.*, 2012). First reports **(Figure 2.23)** of alien fungi have increased consistently since the mid-1800s (Bebber *et al.*, 2013; Fisher *et al.*, 2012; Monteiro *et al.*, 2020; Santini *et al.*, 2013), with approximately 25 per cent of all dated records reported since 2000 (Monteiro *et al.*, 2020). New species discovery for fungi has risen from 1,000-1,500 per year in the mid-2000s, to a peak of more than 2,500 species in 2016 and over 2,000 new species discovered in 2019 (Cheek *et al.*, 2020). Nonetheless, reports of new occurrences are almost certainly underestimated (Bebber *et al.*, 2019). In addition, with rising temperatures and more frequent extreme weather events, fungi are not only able to invade novel geographical areas, but some potentially pathogenic species are also beginning to evolve levels of thermotolerance that could allow them to breach the thermal barriers that have long protected mammals from fungal infections, representing a further threat to human health and wellbeing (Nnadi & Carter, 2021).

Status

Fungi are widely dispersed by humans, often unintentionally or as stowaways, *via* transport through the trade of goods such as plants, seed, wood, shipping containers and other materials (Desprez-Loustau, 2009). Fungi are also dispersed across long and short distances in the atmosphere by wind or water and weather disruptions can play a significant role in spreading fungi into new regions (Anderson *et al.*, 2004; J. K. M. Brown & Hovmøller, 2002). Fungi are being recorded on all continents, including Antarctica **(Figure 2.23)**.

The fungi comprise an immensely diverse biological kingdom that forms complex interactions at multiple ecological levels. Their inconspicuous nature and dispersal

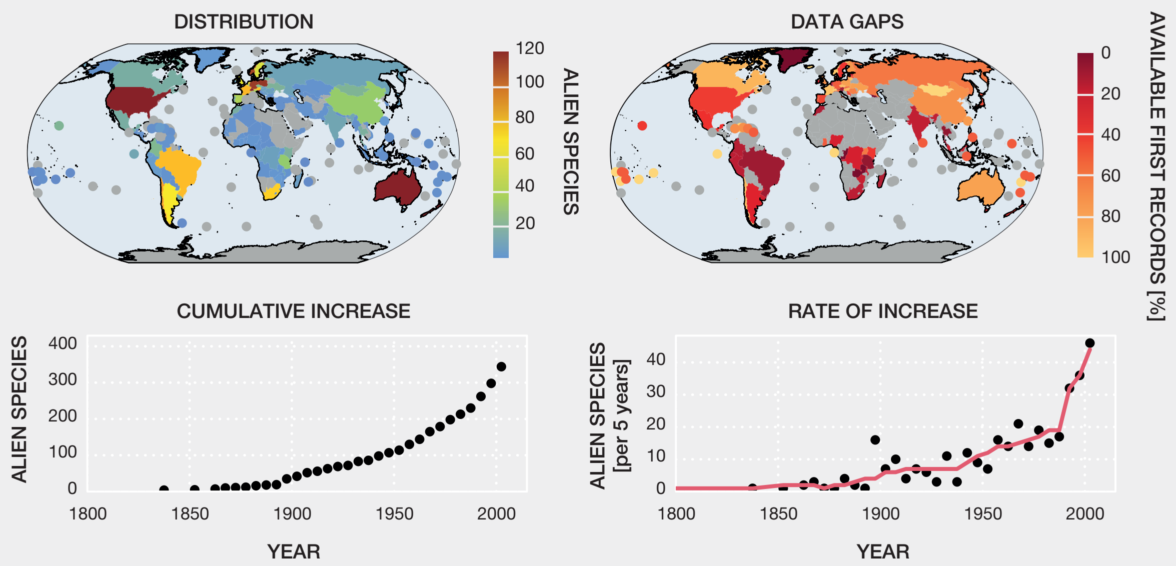

Figure 2 23 **Status, trends, and data gaps for established alien fungi.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note that presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Table 2 17 **Top 10 most widespread invasive alien fungi worldwide.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien species rather than their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
***Ophiostoma novo-ulmi* (Dutch elm disease)**	10	*Ophiostoma ulmi* (Dutch elm disease)	4
***Batrachochytrium dendrobatidis* (chytrid fungus)**	9	*Erysiphe alphitoides* (oak mildew)	3
***Cryphonectria parasitica* (blight of chestnut)**	5	*Melampsoridium hiratsukanum* (alder rust)	3
***Hymenoscyphus fraxineus* (ash dieback)**	5	*Clathrus archeri* (devil's fingers)	2
Pyrrhoderma noxium	5	*Cronartium ribicola* (white pine blister rust)	2

by small, often long-lived spores make the spread of fungi to new locations difficult to control and easy to overlook. Fungal size, particularly the size of the fungal spore-bearing structures, greatly influences how invasive alien fungi are recognized and studied (Desprez-Loustau *et al.*, 2010). The "microfungi," so called because their spore-bearing structures are microscopic, are the most important fungi associated with plant diseases. In contrast, the "macrofungi", which produce large and sometimes vividly coloured spore-bearing structures (e.g., mushrooms), are mostly saprophytes and ectomycorrhizal fungi. Although the distinction between macro and microfungi is artificial, fungal size alone does influence the assessment of invasion dynamics of invasive alien fungi.

About 650 species of macrofungi have been recorded outside their native ranges (Monteiro *et al.*, 2020). Most belong to the orders Agaricales (44 per cent) and Boletales (29 per cent); slightly more than half are ectomycorrhizal, and the remainder are saprotrophic (Monteiro *et al.*, 2020). The most widely distributed alien macrofungi include *Amanita muscaria* (fly agaric), *Amanita phalloides* (death cap), *Phellinus noxius* (brown tea root disease), *Suillus granulatus* (weeping bolete mushroom), and *Suillus luteus* (ectomycorrhizal fungus of pine) (Monteiro *et al.*, 2020). The highest known diversity of macrofungal alien species is in the Southern Hemisphere in countries such as Argentina, Brazil, Chile, New Zealand, and South Africa, and in several European countries, including France, Germany, and the United Kingdom (Monteiro *et al.*, 2020; Vellinga *et al.*, 2009).

Invasive alien fungal symbionts have been co-introduced with their hosts, as in the case of the ectomycorrhizal fungus *Amanita phalloides* (death cap), a native of Europe introduced to Australia and North and South America, probably in soils as consequence of the plant trade (A. Pringle *et al.*, 2009; Vellinga *et al.*, 2009; A. Pringle & Vellinga, 2006). According to Vellinga *et al.* (2009), about 200 species of ectomycorrhizal fungi (including ascomycetes and basidiomycetes) have been introduced into novel habitats due to the transport of *Eucalyptus* and *Pinus* spp. (Pine).

Dung fungi that have accompanied their herbivore partners introduced to the Caribbean islands are a good example (M. J. Richardson, 2008). Commercial use of "biofertilizers" based on arbuscular mycorrhizal fungi is another example. This has led to a global spread of these species (Thomsen & Hart, 2018). Although they can have long-term effects on ecosystems, such alien species tend to go unnoticed (Velásquez *et al.*, 2018) or, in the case of "biofertilizers", unrecognized as an invasion. Some unnoticed alien fungal species may be mutualists associated with only one symbiont species, for example as a plant endobiont. If that symbiont is itself an invasive alien species, a case can be made that the unnoticed mutualist too is behaving invasively by contributing to the success of its associated invasive alien plant. Therefore, an as yet unknown number of additional fungal invasive alien species may remain undetected.

Most parasitic fungi affect plants (Anderson *et al.*, 2004). Examples of invasive alien species include *Cryphonectria parasitica* (blight of chestnut; Gruenwald, 2012), *Ophiostoma* spp. including *Ophiostoma novo-ulmi* (Dutch elm disease; Brasier & Kirk, 2000), *Cronartium ribicola* (white pine blister rust), *Austropuccinia psidii* (myrtle rust), and *Discula destructiva* (dogwood anthracnose). More aggressive genotypes of known plant pathogenic fungi may also arrive as alien species and later become invasive (Arenz *et al.*, 2011). Also important are invasive alien oomycetes such as *Phytophthora pinifolia* causing needle disease in *Pinus radiata* (radiata pine) in Chile (Durán *et al.*, 2008) and hybridization of oomycetes in the genus *Phytophthora* that can cause serious damage to agriculture, horticulture, and forestry (Érsek & Nagy, 2008).

Alien and invasive alien fungi that are pathogenic to animals include *Batrachochytrium dendrobatidis* (chytrid fungi) and *Batrachochytrium salamandrivorans* (chytrid fungi) which are the agents of chytridiomycosis, a disease spread by trade and causing massive global amphibian declines (Berger *et al.*, 2016; Weldon *et al.*, 2004), and *Pseudogymnoascus destructans* (white-nose syndrome fungus) in bats (Hendrix & Bohlen, 2002; Hovmøller *et al.*, 2016; Sikes *et al.*, 2018; Thakur *et al.*, 2019).

2.3.3.2 Chromista, bacteria, protozoans, and viruses

Chromista and other eukaryotic protists constitute several biological kingdoms independent of those for animals, fungi, and plants. Their underlying phylogeny remains poorly understood, with classifications frequently and often radically changing as molecular evidence becomes available. Chromista includes major groups of ecologically highly significant organisms, including many marine algae, diatoms and oomycetes. Note that some groups of Chromista, which are usually considered algae, are addressed in section "Algae" (**section 2.2.2.3**). Here, taxa of the groups Oomycota, Actinobacteria, Chlamydiae, Cyanobacteria, Firmicutes, Proteobacteria and viruses are included.

Along with the true fungi, the Oomycota (with few exceptions including Phytophthora) have rarely been analysed within the context of biological invasions. Recent advances in molecular analyses, however, have shown that at least some of these species have defined natural distributions and can be considered alien if introduced by humans beyond the native range. The emergence of microbial invasive alien species, pathogenic or not, is thus a global phenomenon and a major threat in invasion ecology

(Jack *et al.*, 2021; Litchman, 2010; Mawarda *et al.*, 2020; Ricciardi *et al.*, 2017; Thakur *et al.*, 2019).

Trends

The numbers of alien oomycetes have risen continuously since 1900 (**Figure 2.24**; Santini *et al.*, 2013), as has the numbers for other alien microorganisms as well **(Figure 2.25)**. The new arrivals include some species which are causal agents of serious plant diseases (Blehert *et al.*, 2009; Fisher *et al.*, 2009; Robert *et al.*, 2012; Singh *et al.*, 2008). Global trade is a major driver of oomycete invasions as they are usually unintentionally introduced on their hosts or as contaminants of goods (Sikes *et al.*, 2018). In particular, plants transported with intact root systems, and particularly with soil, are likely to host potentially alien oomycete species, both beneficial and pathogenic.

Historically, there have been several oomycete invasions that have had huge impacts on humans. The most prominent is *Phytophthora infestans* (Phytophthora blight) introduced in the 1800s from North America to Europe. The dispersal of *Phytophthora infestans* is well documented with multiple periods of intense spread over the past 200 years (Fry, 2008). It was the main cause of repeated total potato crop failures resulting in massive famines with millions of deaths and a huge wave of emigration by hundreds of thousands of Europeans (Woodham-Smith, 1962; Yoshida *et al.*, 2013). Importantly, *Phytophthora* species can hybridize, attain greater vigour, and potentially infect a wider host range relative to parent species thereby creating a serious threat to managed and natural systems (Van Poucke *et al.*, 2021).

Status

Well-documented microbial invaders are typically pathogenic organisms which are detected because of their devastating impacts. Anderson *et al.* (2004) provided a list of emerging infectious diseases including *Phytophthora ramorum* (sudden oak death; Gruenwald, 2012).

Biological invasions caused by viruses are also extremely relevant in the context of plants as they account for almost 50 per cent of their emerging infectious diseases (Anderson *et al.*, 2004). In many cases they are transmitted by an invasive alien host species such as *Bemisia tabaci* (tocacco whitefly), which can transmit over 114 virus species (D. R. Jones, 2003). Despite its tropical origin, there have been outbreaks of *Ralstonia solanacearum* biovar 2 (brown potato rot) in Europe where it survives the winter in waterways in association with endemic plants (Stevens & van Elsas, 2010). Many pathogenic microbes are thought to be alien species in the areas in which they were found (Rúa *et al.*, 2011).

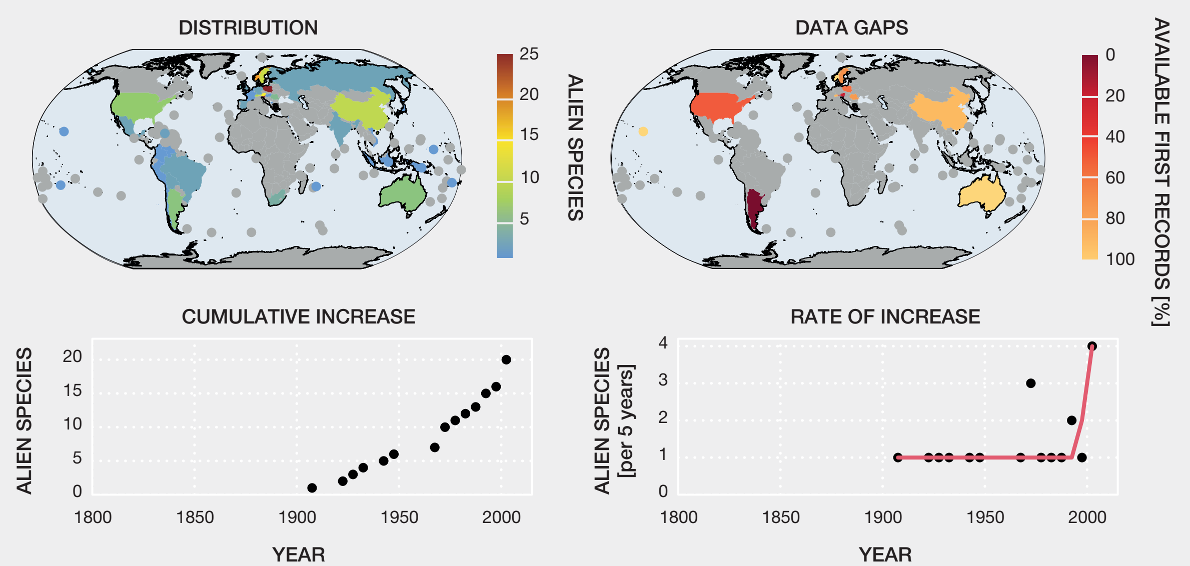

Figure 2 24 **Status, trends, and data gaps for established alien oomycetes.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Detection of non-pathogenic microbial species is more difficult because their impacts can be more subtle and do not result in mortality or disease and are therefore harder to quantify unless previously identified impacts are specifically looked for. Co-invasion of non-pathogenic microbes with plants has been detected in California, United States where genomic analyses revealed that *Ensifer medicae*, a bacterial symbiont associated with the legume *Medicago polymorpha* (bur clover), was introduced from Europe (Porter *et al.*, 2018). Similarly, colonization of New Zealand

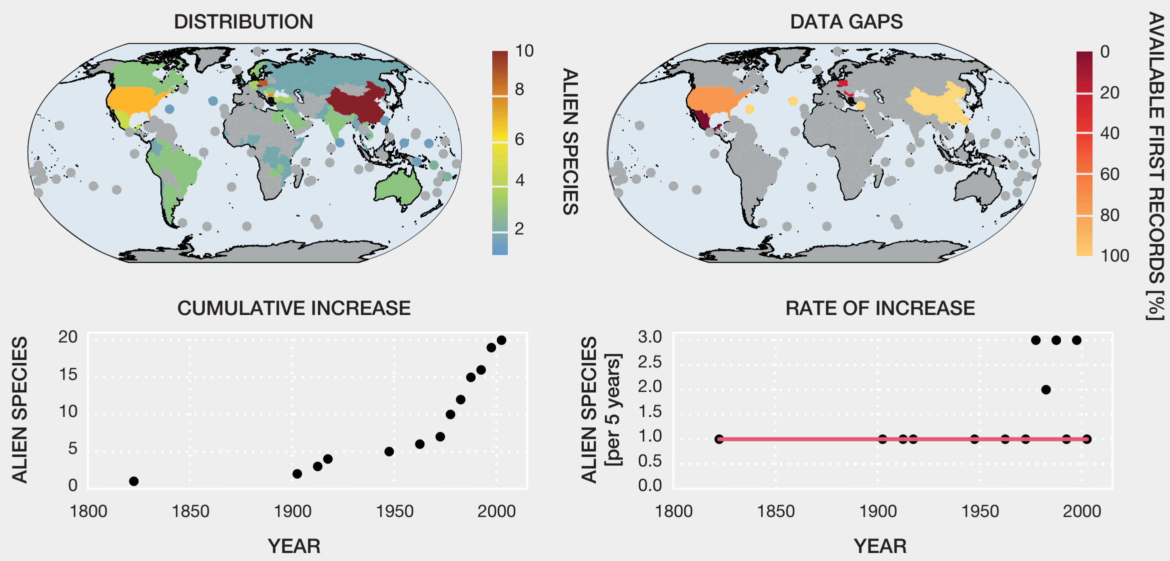

Figure 2 25 **Status, trends, and data gaps for established alien Chromista, bacteria, protozoans, and viruses.**

The number of established alien species per region (upper left) and the amount of available data (upper right) are indicated by colour. The amount of available data is estimated by the proportion of available first records among all records available for that region (**section 2.1.4** for further details). Grey regions denote areas with lacking data. Oceans are tinted for visualization and do not indicate species numbers. Trends are shown in lower panels for cumulative numbers and as a rate of increase (i.e., numbers of established alien species per five years). Smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Table 2 18 **Top 10 most widespread invasive alien taxa of the groups Chromista and bacteria worldwide.**

The number of regions where the respective species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note that this table only refers to the distribution of invasive alien species rather than their impacts which are covered in **Chapter 4** (see **section 2.1.4** for further details on data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species	No. of regions	Species	No. of regions
Vibrio cholerae **(cholera)**	17	*Phytophthora cambivora* (root rot of forest trees)	3
Aphanomyces astaci **(crayfish plague)**	13	*Phytophthora cactorum* (apple collar rot)	2
Phytophthora cinnamomi **(Phytophthora dieback)**	5	*Phytophthora gonapodyides* (oomycetes)	2
Phytophthora ramorum **(sudden oak death)**	4	*Phytophthora infestans* (Phytophthora blight)	2
Yersinia pestis **(black death)**	4	*Phytophthora plurivora* (oomycetes)	2

by European *Lotus corniculatus* (bird's-foot trefoil) coincides with the introduction of its symbiotic partner, the bacterium *Mesorhizobium loti* (Sullivan *et al.*, 1995, 1996).

In most cases, it is unknown whether these introductions spread to other hosts in the introduced habitats which might potentially lead to the displacement of native symbiotic species. Although most known microbial introductions have been reported from Europe, South America, Australia, and New Zealand, these data might be biased by the number of papers published from each country (Vellinga *et al.*, 2009). **Table 2.18** lists the 10 most widespread invasive alien Chromista and bacteria and the number of regions each has invaded.

2.3.3.3 Data and knowledge gaps

Data and knowledge gaps for fungi are vast. Fungi are frequently unnoticed or unreported, particularly in regions where scientific infrastructure is minimal (Desprez-Loustau *et al.*, 2010). Information about alien fungi in different regions can vary tremendously, with biases associated with available scientific infrastructure, taxonomic expertise, crop production, and trade routes (Desprez-Loustau *et al.*, 2010; Lofgren & Stajich, 2021). There are generally far fewer records of fungi than for animals and plants, even from areas with a strong tradition of fieldwork. There are several estimates of the total number of fungal species, with values ranging from 2.2 to 5.1 million, to as many as 11.7 to 13.2 million species (Lofgren & Stajich, 2021). These millions of predicted fungal species greatly eclipse the 146,155 species that are so far discovered and named (Kirk, 2021) and indicate that as many as 98.8 per cent of all fungal species await discovery. Although the rate of new species discoveries has accelerated since the advent of DNA technologies, at the current rate of about 2,000 new fungal species described each year (Cheek *et al.*, 2020), it will be at least a thousand years before a comprehensive inventory of fungal diversity is made.

The continued paucity of rapidly accessible and reliable information for fungi remains a major hurdle for identifying new fungal invasive alien species, particularly cryptogenic fungi, as their initial establishment phase, which is the only stage at which effective countermeasures are feasible, often remains unnoticed until major damage is done (McMullan *et al.*, 2018). Another important knowledge gap is an insufficient understanding of the taxonomic limits of fungal species. This hinders effective quarantine of animal and plant pathogens. Using molecular phylogenetics, several disease-causing microfungi were found to belong to species complexes, and incorrect identifications have led to confusion (Coleman, 2016; X. Lin & Heitman, 2006; Thines & Choi, 2016).

As with fungi, only 10 per cent of all probable oomycete species are estimated to be known and described (Thines, 2014), a large knowledge gap. Information about non-terrestrial species is similarly limited, although several invasions by aquatic algae have been documented (Acosta *et al.*, 2015), including the *Prymnesium parvum* (golden algae) which has successfully established in freshwater ecosystems in several locations in the United States (Roelke *et al.*, 2016; see also **section 2.2.2** including Algae).

Box 2 3 **Evolution during biological invasions.**

Biological invasions have been instrumental in demonstrating that evolution can be rapid enough to contribute to contemporary ecological dynamics and that feedback between ecology and evolution can occur within a few generations (so-called "eco-evolutionary dynamics"; Carroll *et al.*, 2007; Hendry, 2020). Evolution can influence the trends and status of biological invasions by enhancing dispersal rates that lead to species range expansion, improving alien species' performance, and increasing adaptation to novel environments (Suarez & Tsutsui, 2008; Vellend *et al.*, 2007). Indeed, approximately half of the investigated plants and animals show increased size and fecundity in their new range (Parker *et al.*, 2013); many of these differences are likely to have a genetic basis. Adaptive evolution (i.e., evolutionary changes that increase the chance of survival and reproduction) is thought to be common for alien species, especially alien plants (Hodgins *et al.*, 2009). A well-known animal example is *Rhinella marina* (cane toad), which has evolved longer legs and faster movement as its alien range has expanded across Australia (Phillips *et al.*, 2006).

Observations of evolution during invasion initially presented researchers with a paradox. Newly introduced populations tend to be small and are therefore expected to contain low genetic diversity, thereby limiting the population's ability to respond to selection (Sakai *et al.*, 2001). However, some populations that undergo founder effects and genetic bottlenecks can evolve rapidly (Dlugosch & Parker, 2008). In fact, low genetic variation can facilitate invasive behaviour. For example, loss of genetic variation may have reduced intraspecific aggression among alien populations of *Linepithema humile* (Argentine ant), leading to the formation of competitively dominant "supercolonies" (Tsutsui *et al.*, 2000). Other successful invasive alien species have been introduced multiple times and in high numbers (i.e., high propagule pressure), offsetting founder effects and limiting genetic bottlenecks (Roman & Darling, 2007). Indeed, introductions of individuals from different parts of a species' native range can create genetic admixtures (a mixture of previously distinct genetic lineages), boosting levels of standing genetic variation

Box 2 3

in the new range (Meyerson & Cronin, 2013) and potentially providing fitness advantages through hybrid vigour and increased variation, on which selection can act (S. R. Keller & Taylor, 2010). The contribution of novel mutations in large invasive alien populations also cannot be discounted (Colautti & Lau, 2015).

Hybridization and introgression

Genetic variation can also be enhanced during invasion by hybridization among species and interbreeding between native and introduced genotypes (Meyerson *et al.*, 2010; Meyerson & Cronin, 2013); these mechanisms occur commonly and can play an important role during invasion (Hovick & Whitney, 2014; Largiadèr, 2008). Hybridization can facilitate successful invasions if it is beneficial and increases fitness (Bossdorf *et al.*, 2005; Ellstrand & Schierenbeck, 2000; Meyerson *et al.*, 2010; Rius & Darling, 2014); and may help a species overcome Allee effects associated with small sizes of introduced populations (Yamaguchi *et al.*, 2019). For example, hybridization between *Sporobolus alterniflorus* (smooth cordgrass), which was deliberately introduced to the North American Pacific coast from its Atlantic-coast native range, and native *Sporobolus foliosus* (California cordgrass) have generated highly invasive hybrid populations (Daehler & Strong, 1997). Particularly in plants, polyploidy (i.e., genome duplication), sometimes in association with hybridization (Strong & Ayres, 2013), is linked with the success of some alien species through several mechanisms, including enhanced genetic variability (Suda *et al.*, 2015; te Beest *et al.*, 2011). Nonetheless, how frequently the benefits of hybridization outweigh the negative effects is still poorly understood (Hodgins *et al.*, 2018).

Plasticity and adaptation

Invasive alien populations with low genetic variation can also respond to environmental variation in a new range through phenotypic plasticity (Torchyk & Jeschke, 2018). Through plasticity, a single genotype can undergo physiological, phenological, and morphologic changes in response to environmental conditions, which can have significant evolutionary implications (Schlichting, 1986). While it is expected that plasticity will support the establishment and spread of alien species introduced to novel environments (Richards *et al.*, 2006), support for the hypothesis that invasive alien species display greater plasticity than native or non-invasive alien species is mixed (A. M. Davidson *et al.*, 2011; Meyerson *et al.*, 2020; Palacio-López & Gianoli, 2011; Torchyk & Jeschke, 2018). Phenotypic variation can also be generated during invasions through epigenetic mechanisms, that is heritable DNA modifications without changes in the genetic code (Bossdorf *et al.*, 2008). While epigenetic variation has been associated with some successful invasions (C. Liu *et al.*, 2020; Richards *et al.*, 2012), it is too early to generalize about the importance of this mechanism for invasions (Bock *et al.*, 2015). Invasive alien species can also adapt to environmental conditions in their new range and increase their abundance, though few empirical studies have quantified these links (Hodgins *et al.*, 2018). For example, *Lythrum salicaria* (purple loosestrife) in North America has experienced demographic benefits of adaptation estimated to be equivalent to those that the species enjoys from natural enemy release (Colautti & Barrett, 2013).

Data and knowledge gaps

A key uncertainty is how much evolution favours or hinders the outcome of a biological invasion, for example, by making the difference between invasion success and failure (Bock *et al.*, 2015). To this end, perspectives from ecology and evolution could be further integrated by combining genomic tools with more classical experimental and comparative studies to test the mechanisms and consequences of evolution during invasion (Holman *et al.*, 2019; McCartney *et al.*, 2019). Another critical question is to what extent evolution allows alien species to colonize environments that are outside of their native-range ecological niches (Moran & Alexander, 2014; Pearman *et al.*, 2008). Settling this question is important for commonly used tools such as species distribution models to forecast potential distributions of alien species (Pearman *et al.*, 2008). Finally, studies of invasions have shown that some species can rapidly adapt to changing environments (Colautti & Lau, 2015; Hodgins *et al.*, 2018). Alien species may be exceptionally responsive to interacting global-change drivers (Moran & Alexander, 2014), such as climate change or land-use change, a topic warranting further research (**Chapter 3**, **sections 3.5** and **3.6.1**).

Linking evolution and molecular tools to invasive alien species impacts and management

Just as alien species adapt to their novel environments, so too have native species evolved in response to the novel selection pressures posed by alien species. Evolutionary responses to exposure to alien competitors appear to be widespread in plants (Oduor, 2013). Thus, evolution may partially mitigate the negative impacts of invasive alien species on native communities (Carroll, 2011). This understanding also points to ways in which genetic tools and evolutionary principles may help to mitigate some of the impacts of invasive alien species (Chown *et al.*, 2015; Lankau *et al.*, 2011).

Information about the evolutionary/phylogeographic history of alien species obtained by using molecular markers and up-to-date statistical methods can also have several practical benefits for alien species monitoring and management (Lankau *et al.*, 2011). Such knowledge can improve the efficacy of biocontrol programmes by targeting biocontrol agents from within the source region of a given invasive alien species (Chown *et al.*, 2015) and provide better delimitation of source regions and introduction pathways, which can be obtained using high-resolution genomic tools (Hudson *et al.*, 2021, 2022). While it is widely recognized that biological

Box 2 3

invasions constitute a natural experimental framework for the study of contemporary evolution, a good understanding of source regions and introduction pathways (i.e., routes of invasion/introduction) is essential. Knowledge of those routes makes it possible to precisely compare introduced populations to their original source population(s) and thus determine whether the invaders have, for example, undergone an adaptive change that has favoured them in their new living environment. This change may result from the selection of genetic variants that are rare in the original source population(s) but favoured in the new environment. The reconstruction of routes of invasion/introduction is, therefore, crucial to define and test different hypotheses concerning the environmental and evolutionary factors underlying biological invasions and their success (Estoup & Guillemaud, 2010; S. R. Keller & Taylor, 2008). Bulk screening by using metabarcoding approaches may be used to flag recognized invaders at ports of entry and so prevent the introduction of harmful species (or new genotypes of already introduced species). The potential for molecular instruments to detect the spread of invasive alien species is important, although many challenges remain (Handley, 2015).

2.4 TRENDS AND STATUS OF ALIEN AND INVASIVE ALIEN SPECIES BY IPBES REGIONS

This section reports on the temporal trends and status of the distribution of alien and invasive alien species across IPBES regions (**section 2.4.1**), and for the individual IPBES regions Africa (**section 2.4.2**), the Americas (**section 2.4.3**), Asia and the Pacific (**section 2.4.4**), and Europe and Central Asia (**section 2.4.5**), and their respective sub-regions. A description of IPBES regions and sub-regions including a spatial representation is provided online (IPBES Technical Support Unit On Knowledge And Data, 2021) and in **Chapter 1**, **section 1.6.4**. For each IPBES region, dynamics on islands and data and knowledge gaps are provided as well. A global synthesis on the dynamics on islands and in protected areas is provided in boxes (**Boxes 2.4** and **2.5**).

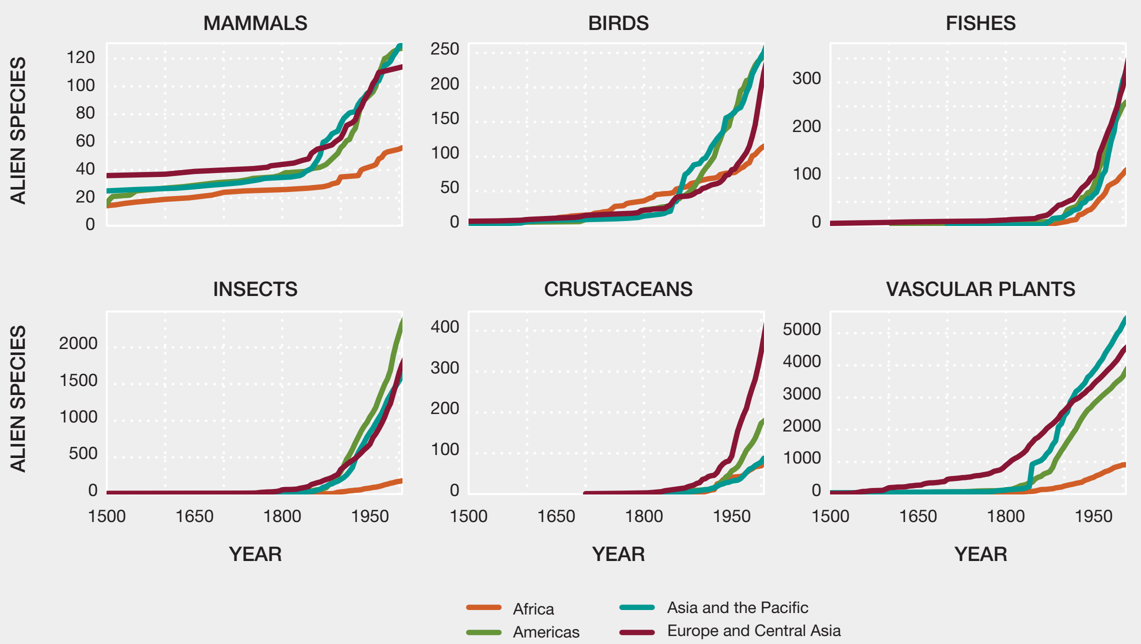

Figure 2 26 **Trends in numbers of established alien species across IPBES regions.**

The panels show cumulative numbers of established alien species for different taxonomic groups. Numbers shown underestimate the actual extent of established alien species occurrences due to a lack of data (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

2.4.1 Overview of trends and status by IPBES regions

Trends

The number of established alien species records has increased for all taxonomic groups and for all IPBES regions since 1500 with particularly steep escalations observed after 1800 **(Figure 2.26)**. Before 1800, the number of records is particularly low for insects and crustaceans. However, this is likely because of the lack of data, which is particularly common for invertebrate groups (**section 2.3.1.11**). Likewise, the comparatively high numbers of established alien species observed for Europe and Central Asia is likely influenced by the higher availability of records for Europe and biases in the underlying database. Nonetheless, no saturation of established alien species is observed for any region (Seebens, Essl, *et al.*, 2017).

Status

Across taxonomic groups, vascular plants provide the by far largest contribution to global established alien species numbers, followed by insects and fishes **(Table 2.19)**. For many taxonomic groups, all IPBES regions except Africa report similar numbers of established alien species **(Table 2.19)**. For instance, the numbers of alien vascular plant species reported for the Americas, Asia and the Pacific, Europe and Central Asia are comparable in their range, while the numbers for Africa are much lower. Similar patterns are observed for alien bird and fish species. On the other hand, algae show a different pattern with Europe and Central Asia harbouring the highest established alien species numbers, followed by the Americas, Asia and the Pacific, and Africa. However, this pattern may be influenced by variation in research intensity around the world. **Box 2.6** also presents an overview of alien and invasive alien species on land managed by Indigenous Peoples and local communities.

Table 2 19 **Numbers of established alien species across IPBES regions.**

Numbers of established alien species can vary depending on data sources. For mammals, birds, and vascular plants, ranges of values indicate variation among databases (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this table is available at https://doi.org/10.5281/zenodo.7615582

	Africa	Americas	Asia and the Pacific	Europe and Central Asia	Total
Mammals	30-80	83-164	97-163	72-164	197-368
Birds	121-133	249-287	287-336	221-630	495-877
Fishes	187	803	633	469	1,451
Reptiles	158	192	103	98	411
Amphibians	12	62	43	43	135
Insects	344	2,636	2,017	2,747	6,795
Arachnids	94	207	129	289	500
Molluscs	142	255	261	584	826
Crustaceans	111	213	149	451	813
Vascular plants	3,109-4,498	8,005-9,325	6,141-9,101	5,146-8,519	13,081-18,543
Algae	58	193	157	526	734
Bryophytes	0	48	32	23	88
Fungi	122	363	363	609	1,149
Oomycetes	4	12	12	59	70
Bacteria and protozoans	4	14	12	23	38
Total	5,033-6,484	14,853-16,292	11,722-14,797	13,754-17,628	26,783-32,798

Box 2 4 Protected areas: A global assessment of trends and status of alien and invasive alien species.

Protected areas around the world are crucial for preserving and sustaining biodiversity, ecosystem processes and human well-being (Gaston *et al.*, 2008; Naughton-Treves *et al.*, 2005). Increasingly, these areas are being threatened by numerous drivers of change in nature that are challenging the effective management of over 200 thousand protected areas globally (Osipova *et al.*, 2017; UNEP-WCMC *et al.*, 2021). Biological invasions constitute a major threat to protected areas (Goodman, 2003; Osipova *et al.*, 2017; Pyšek, Hulme, *et al.*, 2020; Schulze *et al.*, 2018), a concern that dates back to the 1860s (Foxcroft *et al.*, 2017).

Seminal work on invasions in terrestrial protected areas carried out during the Scientific Committee on Problems of the Environment (SCOPE) project in the 1980s found that all 24 studied terrestrial protected areas faced challenges from invasive alien species and that invasions were not only an issue within disturbed sites (Mooney *et al.*, 2005; Usher, 1988), but also in relatively undisturbed nature reserves. The SCOPE report also found that islands faced higher threats than mainland areas, that there was an inverse relationship between protected area size and the number of introduced species in arid land and chaparral biomes, and that there was positive correlation between number of human visitors and the presence of invasive alien species (Usher, 1988). In a study that revisited 21 of the originally studied protected areas and compared how the status of biological invasions has changed over the last 30 years, Shackleton *et al.* (2020) found that of all the taxa analyzed, invasive plants pose the greatest continued threat, and their numbers have increased in 31 per cent of the protected areas. Mammal invasions now represent a lesser threat due to effective management in many protected areas, with fewer invasive alien mammals now listed in 43per cent of protected areas. Invasions by amphibians, reptiles, and fish have remained fairly stable over the past three decades (R. T. Shackleton, Foxcroft, *et al.*, 2020). The limited number of study sites included were biased towards mainland United States and Africa making regional comparisons and trends hard to meaningfully assess. More comprehensive global assessments using similar methods would address a major knowledge gap and better evaluate status and change globally providing important information for international policy (**Glossary**) mandates.

The subsequent uptake of coordinated global academic projects on protected areas has been limited, particularly for marine systems leaving many knowledge gaps on the status of invasive alien species in protected areas and the broad-scale status trends. According to Shackelton *et al.* (2020) there is a lack of data on freshwater invertebrates, marine species, and other taxa creating a taxonomic bias in invasion science. However, some review and synthesis work (e.g., Foxcroft *et al.*, 2013, 2017; X. Liu *et al.*, 2020; R. T. Shackleton, Bertzky, *et al.*, 2020; R. T. Shackleton, Foxcroft, *et al.*, 2020; see above) has strengthened information on the current status and key trends of invasive alien species in protected areas globally, but each effort has limitations and greater coordination on taxa and management is needed.

In "Plant invasion in Protected Areas", Foxcroft *et al.* (2013) identified and illustrated key impacts of invasive alien species and outlined some mechanisms of invasion in protected areas and contributed to assessing management interventions, helping to synthesize and outline both the status of invasive alien species in protected areas and key knowledge gaps. Drawing on 14 case studies from around the world that included information from over 135 protected areas globally, the authors detailed assessments and baseline information and elucidated regional patterns and threats. One surprising result was that while intentional introductions of invasive alien species into protected areas have been assumed to be low, this is not the case. This point is further supported by Foxcroft *et al.* (2008) and Toral-Granda *et al.* (2017). Authors show that even Arctic regions now face challenges from invasive alien species (Shaw, 2013). Very few protected areas globally have good baseline information and only a handful of well-studied protected areas have robust invasive alien species lists available. Regionally there are also large differences in monitoring and information. The United States, Oceania, and some parts of Europe have more information than other regions. For example, J. A. Allen *et al.* (2009) highlight that there are over 7.3 million ha of invasions in 218 protected areas in the United States, with over 20,300 distinct invasion clusters by over 3,750 invasive alien species. In Central and Western Europe, Braun *et al.* (2016) collected and collated data on 53 invasive plant species in 46 large, protected areas finding that in 86 per cent of protected areas at least one of the 46 target invasive plants was present, and that 80 per cent of protected areas did conduct some form of management. The mean number of invasive plants was 11.2 per protected area, however, most of them only managed a mean 4.3 species accounting for around 3 per cent of park budgets. Interestingly, park size and age had no effect on invasive alien species presence or management.

A review on plant invasion science research in protected areas (Foxcroft *et al.*, 2017) yielded some important information on trends and status highlighting key advances in invasion science in protected areas, important policies starting with the Convention Relative to the Preservation of Fauna and Flora in their Natural State in 1933, the twelfth meeting of the Conference of the Contracting Parties to the Ramsar Convention on Wetlands in 2015, and 13 other important policy support mechanisms in-between. This review also identified 59 of the most common invasive plants in protected areas: eight species (*Arundo donax* (giant reed), *Pontederia crassipes* (water hyacinth), *Lantana camara* (lantana), *Melia azedarach* (Chinaberry), *Poa annua* (annual meadowgrass), *Psidium guajava* (guava), *Robinia pseudoacacia* (black locust), and *Rumex acetosella* (sheep's sorrel)) occur in more than

Box 2 4

150 protected areas globally. The review showed that North America and Europe dominate work on plant invasions in protected areas globally, followed by Africa and Oceania, with very limited knowledge from other world regions, particularly in South America and Asia.

More recently, key syntheses have assessed the trends and status of invasions in terrestrial and inland waters protected areas globally (e.g., X. Liu *et al.*, 2020; R. T. Shackleton, Bertzky, *et al.*, 2020). X. Liu *et al.* (2020) assessed the establishment of 894 terrestrial alien vertebrates and invertebrates in almost 200 thousand protected areas globally and found that very few (over 10 per cent) of protected areas harbour established alien animals, but the majority (89–99 per cent) have an established population of at least one alien animal species within 10-100 km from their borders. There are 520 alien animal species in protected areas globally, the most common being birds (4.7 per cent of the protected areas, 252 species), followed by mammals (3.7 per cent, 91 species), invertebrates (2.2 per cent, 63 species), amphibians (0.5 per cent, 48 species) and reptiles (0.4 per cent, 66 species) (X. Liu *et al.*, 2020). X. Liu *et al.* (2020) highlight that larger protected areas, those more recently inscribed, and those with a higher protection status were surprisingly more prone to a higher richness of alien animals. Furthermore, X. Liu *et al.* (2020) found that globally, protected areas in some regions and biomes are more at risk from alien animals, including birds, mammals, invertebrates, amphibian and reptiles; particularly in (sub)tropical Pacific and Caribbean Islands and New Zealand, as well as temperate mixed forests, savannas, and grasslands in the United States, western Europe, and Australia. Additionally, X. Liu *et al.* (2020) highlight that Africa and Asia are most often donors of alien animal species with North America and Europe being key recipient areas **(Figure 2.27)**.

Shackleton, Bertzky, *et al.* (2020) assessed the status of biological invasions and their management in 241 natural and mixed World Heritage Sites globally and found that just over half (53 per cent) were explicitely or implicitly reported to be threatened by invasive alien species through formal IUCN/ United Nations Educational, Scientific, and Cultural Organization (UNESCO) monitoring initiatives. It is suspected that this number is much higher. Almost 300 different invasive alien species were reported to be invading World Heritage Sites. However, detailed information through UNESCO and IUCN monitoring programmes yielded limited and inconstant information so broad-scale trends were hard to assess. To overcome this a seven-step monitoring and reporting framework was developed to better collate data moving forward. This includes: (i) evaluating pathways, (ii) compiling inventories of species, (iii) identifying current impacts, (iv) reporting on management, (v) predicting future threats and management needs, (vi) identifying knowledge gaps, and (vii) assigning an overall threat level. This framework could easily be used in all categories of protected areas and could be a priority moving forward to improve monitoring and understanding.

Marine protected areas *"… as oases of biodiversity, serve as the last rampart against these invasive alien species"* (Francour *et al.*, 2010). Alas, this is a wishful premise and biological invasions are having a large impact on marine protected areas worldwide. Large-scale global syntheses on the topic of marine invasions and protected areas are lacking, however, research on certain areas and species has provided important insights which are summarized here. Generally, European oceans and seas, as well as northern Atlantic and Pacific oceans, are most at threat from marine invasive alien species (M. J. Costello *et al.*, 2021). More specifically, 53 marine alien species, nearly all newly reported or newly recognized as introduced, were recently documented in the Galápagos Marine Reserve, which is a large, biologically diverse and remote protected area (Carlton *et al.*, 2019). Surveys of rocky reef fish assemblages conducted since 2000 in Mediterranean marine protected areas showed no differences in invasive fish density and biomass as compared to adjacent unprotected areas. In the south and eastern Mediterranean Sea invasive alien species have higher species richness and biomass as compared to local fish biota (D'Amen & Azzurro, 2020; Galil, 2017; Giakoumi *et al.*, 2019; Guidetti *et al.*, 2014). Indeed, a recent assessment in protected areas along the Mediterranean coast of Turkey identified 289 alien vertebrates, invertebrates and algae (Bilecenoğlu & Çınar, 2021). The reduction of protected areas to nursery sites for certain invasive alien species is most acute in the South-eastern Mediterranean but occurs throughout the sea and in the adjacent Atlantic (Blanco *et al.*, 2020; Cacabelos *et al.*, 2020; Mazaris & Katsanevakis, 2018; Wangensteen *et al.*, 2018). From a species point of view, the spread of the venomous Indo-Pacific lionfish, *Pterois volitans* (red lionfish) and *Pterois miles* (lionfish), across the tropical western Atlantic and the Caribbean Sea was swift, not sparing marine protected areas, including large, established, well-cared for and remote ones (e.g., Florida Keys National Marine Sanctuary, United States; Flower Garden Banks National Marine Sanctuary, United States; The Parque Nacional Arrecife Alacranes, Mexico) (Johnston *et al.*, 2013; López-Gómez *et al.*, 2014; Ruttenberg *et al.*, 2012), illustrating the threat that invasive marine species pose to conservation. Poor management and the lack of effective policies have been nullifying conservation goals in marine protected areas in regions exposed to biological invasions (Bilecenoğlu & Çınar, 2021; B. Galil, 2017; Mazaris & Katsanevakis, 2018; **Chapters 5** and **6**).

Foxcroft *et al.* (2017) mention three key needs to better understand the current status of biological invasions and their management in protected areas globally and to better assess key trends. These include (i) establish a global working group to better coordinate research, (ii) develop standardized protocols and tools for large-scale and long-term monitoring of invasive alien species in protected areas globally, and (iii) better account for and respond to different socioecological contexts in research and management. Importantly, many regions of the world have limited baseline and empirical evidence concerning biological invasions and their management making this fundamental research crucial.

Box 2 4

The collection of baseline data is increasingly being conducted in data poor areas (e.g., Bhatta *et al.*, 2020; Foxcroft *et al.*, 2017; Padmanaba *et al.*, 2017), but more is needed. Furthermore, improved monitoring and assessment globally is important to answer long-standing and disputed questions relating to invasions in protected areas. For example, whether or not protected areas impose biotic resistance (**Glossary**) against invasions (Meiners & Pickett, 2013). Some evidence suggests protected areas act as a barrier, or refuge, against invasions (Ackerman *et al.*, 2017; Foxcroft, Jarošík, *et al.*, 2010; Gallardo *et al.*, 2017), but other studies show the contrary (Byers, 2005; Holenstein *et al.*, 2021; Klinger *et al.*, 2006). Further work drawing on a multitude of taxa in different socioecological systems is needed to fully understand the role of protected areas in invasions, which is likely to differ by taxa and environmental settings.

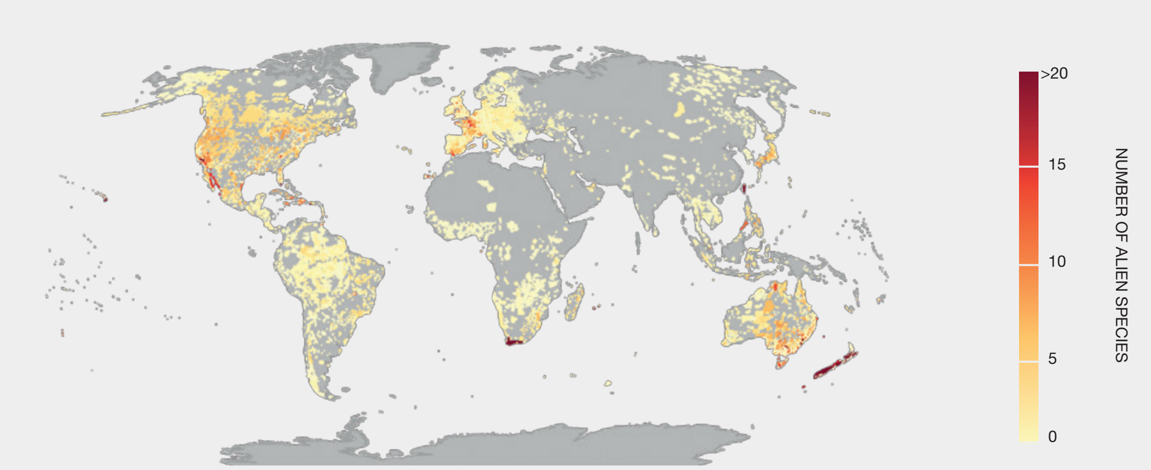

Figure 2 27 **Numbers of established alien vertebrate species per terrestrial protected area.**

Among the top 50 protected areas, 32 per cent are located in New Zealand, 26 per cent in Taiwan, Province of China, 16 per cent in the United States (mostly on Hawaii), 12 per cent in Great Britain and 6 per cent on Réunion. Adapted from X. Liu *et al.* (2020), https://doi.org/10.1038/s41467-020-16719-2, under license CC BY 4.0.

Box 2 5 **Islands: A global assessment of trends and status of alien and invasive alien species.**

One-quarter of the countries in the world are islands or groups of islands, and over two-thirds of all countries include islands (Russell & Kueffer, 2019). Taken together, the Earth's islands represent 5.3 per cent of the total land surface (Global Islands Network, 2021; Tershy *et al.*, 2015). Because of their very high rates of endemism (9.5 and 8.1 times higher than continents for vascular plants and vertebrates, respectively), and with over 20 per cent of the world's terrestrial species, islands are considered centres of biodiversity (Kier *et al.*, 2009). As a result, 10 of the 35 world's biodiversity hotspots (i.e., regions where biodiversity is both the richest and the most threatened (Mittermeier *et al.*, 2011) are entirely, or largely consist of, islands (Bellard *et al.*, 2014). Globally, islands represent concentrated regions of biodiversity loss in the past and present, and this trend is predicted to continue in the future (Russell & Kueffer, 2019; Whittaker & Fernández-Palacios, 2006).

Islands harbour some of the highest numbers of established alien species (Dawson *et al.*, 2017; Essl *et al.*, 2019), particularly small and remote tropical and sub-tropical islands with high numbers of invasive alien plants per unit of surface (Pyšek, Pergl, *et al.*, 2017), a pattern that holds across taxonomic groups (Moser *et al.*, 2018; Turbelin *et al.*, 2017). This is especially acute in former European island colonies with long histories of repeated species introductions (Turbelin *et al.*, 2017). Furthermore, nearly 50 per cent of all species at risk (**Glossary**) of extinction on the IUCN Red List are found on islands and species on islands are more likely to be threatened by biological invasions (almost three-quarters of threatened species; Leclerc *et al.*, 2018). While all threats interact on islands to cause declines in native species abundance, biological invasions consistently lead to the extinction of insular populations, particularly through predation and disease (Russell & Kueffer, 2019; **Chapter 4**, **section 4.3.1.1**). However,

Box 2 5

particularly independent small island developing states (SIDS) and island territories with dependencies on larger continental economies (Blackburn *et al.*, 2016; Meyerson & Reaser, 2003; Reaser & Meyerson, 2003; Russell *et al.*, 2017) have few resources for invasive alien species research, management, cooperation, and capacity-building (Reaser & Meyerson, 2003; Veitch *et al.*, 2019).

Trends

Temporal trends of biological invasions on islands can be classified into three distinct periods with contrasting dynamics; first contact (Indigenous Peoples and local communities), modern history (1500), and the contemporary twentieth century onwards era (Keppel *et al.*, 2014; Russell & Kueffer, 2019; **Figure 2.28**). In the first period, island syndromes (Wroe *et al.*, 2006) and the lack of refugia on small islands made insular species more vulnerable to biological invasions than continental species (Wroe *et al.*, 2006). The second period corresponds to the “Age of Discovery”, the timing of which in different parts of the world coincided with colonization of islands by Europeans (Russell & Kueffer, 2019). During this period, unintentional and intentional (and sometimes repeated) introductions of many animals and plants were facilitated by the establishment of regular shipping lines (Seebens *et al.*, 2013). This led to successful invasions by a large number of species on many islands of various ecosystem types (Russell & Kueffer, 2019). The third period is associated with globalization that included a distinct increase in world trade, migration, and tourism, all of which affected islands worldwide. The emergence and rise of rapid international transit increased substantially both the diversity of introduction vectors and pathways (Hulme, 2009, 2021; Meyerson & Mooney, 2007), and the associated number of these introductions (van Kleunen *et al.*, 2015). The number, frequency, and geographic origin of biological invasions to and among islands also increased with time, following the growth of human populations on these islands (both residents and tourists), as exemplified by the Galapagos (Toral-Granda *et al.*, 2017). At the same time, awareness was rising, and more research was underway to detect and report new species. Other important predictors for established alien species on islands are the existence of military bases or paved airfields (Denslow *et al.*, 2009).

Most introduced species on islands today only occupy a small portion of their final predicted range and are thus likely to expand further (M. J. B. Dyer *et al.*, 2018; Trueman *et al.*, 2010). In addition, more species from both the existing pool of alien species and those species not currently introduced outside their native range will continue to colonize and establish on islands in the future (Bellard *et al.*, 2017). Islands are also disproportionately vulnerable to climate change which may increase the rate of establishment and spread of many invasive alien species on islands (X. Li *et al.*, 2020). More frequent climate-induced disturbances (e.g., flooding, treefall, and landslides caused by tropical cyclones) and/or droughts increase the invasibility of native ecosystems affecting, for instance, the structure of island forests (Boehmer, 2011; Ehbrecht *et al.*, 2021; Pouteau & Birnbaum, 2016; Wyse *et al.*, 2018).

The accumulation rate of established alien species on islands is not slowing and the future invasive alien species will differ in type from species that have invaded islands in the past. These emerging invasive alien species include groups such as microorganisms and pathogens, as well as reptiles from the pet trade (Apanius *et al.*, 2000; Russell & Kueffer, 2019), which will likely lead to new species interactions with both direct and indirect ecological consequences (Forey *et al.*, 2021; J.-Y. Meyer *et al.*, 2021). In the future, the vectors and pathways of biological invasions are predicted to further evolve and to keep interacting with other drivers of change in nature, such as climate change (Russell *et al.*, 2017), and will continue to be of great concern for biodiversity conservation (Lenzner *et al.*, 2020; S. Taylor & Kumar, 2016). For instance, climate-induced forest decline is likely to increase the vulnerability of Pacific Island rainforests to invasive alien plants (Boehmer, 2011; Mertelmeyer *et al.*, 2019) and facilitate invasional meltdowns (Minden *et al.*, 2010).

Status

Most islands are affected by biological invasions with insular ecosystems being the recipients of 80 per cent of documented bird and mammal introductions (Ebenhard, 1988). At least 65 major island groups have been invaded by *Felis catus* (cat) (Atkinson, 1989) and over 80 per cent of all major island groups have also been invaded by *Rattus* spp. (rat) (Atkinson, 1985). If plants and invertebrates are included in assessments, biodiversity is most severely affected by biological invasions in the Pacific and Atlantic insular regions (Leclerc *et al.*, 2018). For plants, 26 per cent (82 islands) of islands covered in the GloNAF database harbour more established alien than native species (Essl *et al.*, 2019). The identity of invasive alien species and their impacts differ by region, island type, and associated ecosystems, but the cumulative pattern of impacts is consistent across world regions (Leclerc *et al.*, 2020).

Across SIDS, 8,668 presence records for 2,034 potential invasive alien species have been registered, 76 per cent of which are plants, 23 per cent animals, and 1 per cent fungi, chromists, viruses, bacteria, and protozoa (Russell *et al.*, 2017). Over half (53 per cent) of these species were identified as invasive alien species on at least one SIDS, while information was often lacking for the remaining species (Lenz *et al.*, 2021). Long-distance transportation by ship and plane dominates invasive alien species pathways to islands, distinguishing islands from continents and natural colonization in rate and type (Hulme *et al.*, 2008), such as for *Anolis* spp. (anole lizards) on Caribbean islands (Helmus *et al.*, 2014). Only one study has focused on plant invasions in urban environments of SIDS (Lowry *et al.*, 2020). Given rapid changes expected in Pacific country urban areas in coming decades, it is a critical to fill this gap (ADB, 2012).

Box 2 5

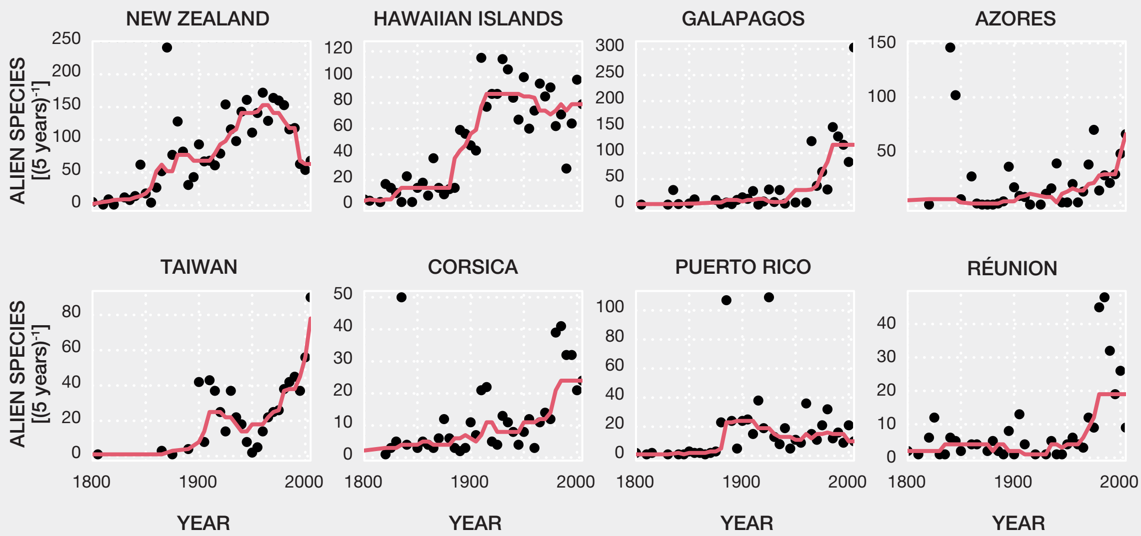

Figure 2 28 **Trends in numbers of established alien species for selected islands.**

The panels show numbers of established alien species per five-year intervals for those islands with the highest numbers of recorded established alien species. Numbers shown underestimate the actual extent of alien species occurrences due to a lack of data. Smoothed trends (lines) are calculated as running medians (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Box 2 6 Land managed, used or owned by Indigenous Peoples and local communities: A global assessment of trends and status of alien and invasive alien species.

Indigenous Peoples and local communities (i.e., typically ethnic groups who are descended from and identify with the original inhabitants of a given region) manage or have tenure rights over a large area of land. For Indigenous Peoples only, it is estimated that they manage or have tenure rights for at least 28 per cent of the total land area worldwide (Garnett *et al.*, 2018). Their land (hereafter called "Indigenous lands") intersects with 40 per cent of the world's protected areas and hosts higher amounts of natural areas compared to other lands (Garnett *et al.*, 2018). Although Indigenous lands are often less inhabited and more remote than other lands, they do not escape anthropogenic pressures. It is unsurprising to find many alien and invasive alien species on lands managed by Indigenous Peoples and local communities and indeed has been frequently reported from such lands all over the world (Gautam *et al.*, 2013; Kannan *et al.*, 2016; Ksenofontov *et al.*, 2019; Miranda-Chumacero *et al.*, 2012; Thorn, 2019). To date, no study has investigated the distribution of alien and invasive alien species on Indigenous lands.

The following analysis was conducted to deepen the understanding about the distribution of alien and invasive alien species on Indigenous land. As described in **section 2.1.4**, occurrences of populations of more than 17,000 established alien species worldwide were obtained using occurrence records provided by GBIF and the Ocean Biodiversity Information System (OBIS). These point-wise occurrences were integrated with a spatial layer of land managed, used or owned by Indigenous Peoples (Garnett *et al.*, 2018) to determine the total number of established alien and invasive alien species recorded on Indigenous lands.

This analysis revealed that, in total, 6,351 established alien species have been recorded on Indigenous lands, which is 34 per cent of all established alien species recorded worldwide in this data set. The number of invasive alien species according to the GRIIS database (Pagad *et al.*, 2022) amounts to 2,355 (56 per cent of the total number globally) on these lands, although it could not be determined whether the invasive alien species pose any impact on these lands (see **Chapter 4**, **section 4.6** for a detailed assessment of impacts by Indigenous Peoples and local communities). The number of established alien species recorded on Indigenous lands is highly correlated with the total number of established alien species of the same

Box 2 6

country (t-test: t=12.8, df=77, $p<0.001$, r=0.82). That is, in countries with high numbers of established alien species, those numbers are also high on Indigenous lands. However, the number of established alien species recorded on Indigenous land is on average consistently lower compared to those numbers recorded on other lands also after taking area into account **(Figure 2.29)**. Hotspots of occurrences with high established alien species numbers on Indigenous lands were found all over the world but particularly in Australia (2,624 alien species), United States (1,719), Mexico (746), Sweden (690) and Russia (650). The same sequence applies to invasive alien species numbers, although at a lower magnitude: Australia (1,172 invasive alien species), United States (691), Mexico (481), Sweden (441), and Russia (436) **(Figure 2.29)**.

An analysis of the trends of alien and invasive alien species on Indigenous lands is currently missing due to a lack of data, but it seems very likely that the number of established alien species on Indigenous lands increased as observed for other regions (**Figures 2.4** and **2.26**) and so are the impacts they cause. A clear knowledge gap exists for information about the trends and status of invasive alien species in coastal waters managed by Indigenous Peoples and local communities.

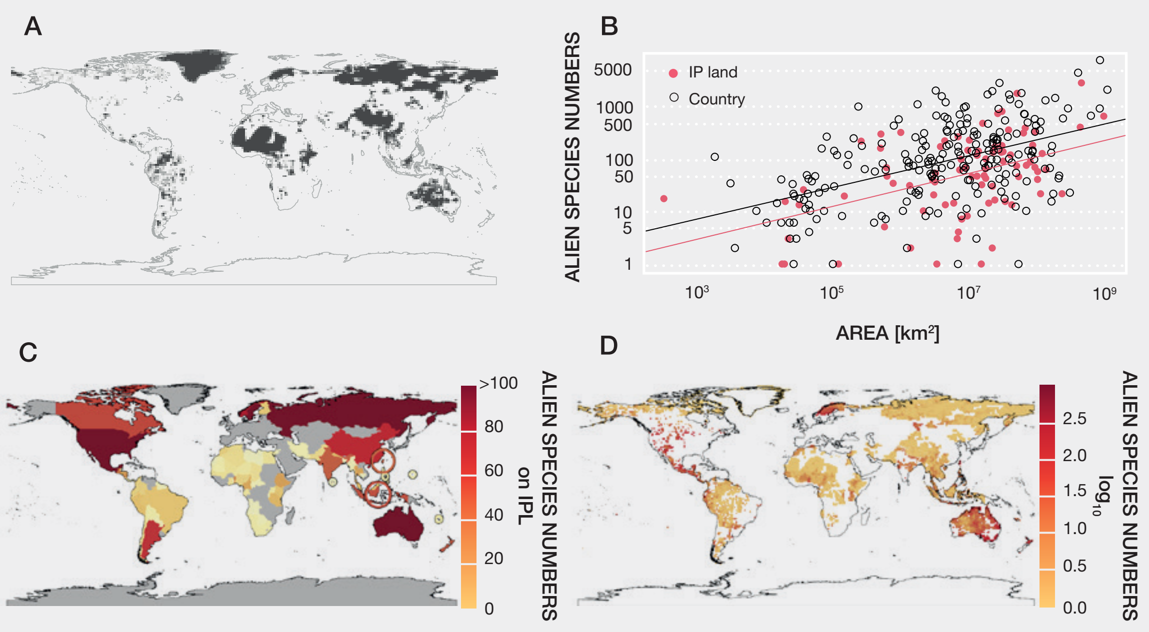

Figure 2 29 **Invasive alien species on Indigenous People's land.**

(A) Land managed, used or owned by Indigenous Peoples. **(B)** Species-area relationships for established alien species per country (circles) and per area of Indigenous lands (IP) lands (dots), showing a consistently lower number of established alien species on Indigenous lands. **(C)** Number of alien species on Indigenous lands per country. **(D)** Number of established alien species on Indigenous lands per grid cell. A data management report for this figure is available at https://doi.org/10.5281/zenodo.7615582

2.4.2 Trends and status of alien and invasive alien species in Africa

This section reports on the trends and status of established alien species of Africa for animals (**section 2.4.2.1**), plants (**section 2.4.2.2**), microorganisms (**section 2.4.2.3**), and islands (**section 2.4.2.4**), and provides an overview of data and knowledge gaps (**section 2.4.2.5**). A description of IPBES regions and sub-regions including a spatial representation is provided online (IPBES Technical Support Unit On Knowledge And Data, 2021) and in **Chapter 1**, **section 1.6.4**.

2.4.2.1 Animals

Trends

The first alien mammal species to arrive in Africa were probably domesticated bovids, pigs, cats, and dogs during the spread of agriculture, followed by commensal rodents, mostly limited at present to anthropized and densely populated areas (Long, 2003). Other introductions took place on the western coast of North Africa where *Mustela nivalis* (weasel) was likely a rodent biocontrol agent, *Apodemus sylvaticus* (long-tailed field mouse), a stowaway, and *Bubalus bubalis* (Asian water buffalo) livestock. More introductions began in the twelfth century such as *Suncus murinus* (Asian house shrew) as a stowaway. A rapid increase of mammal introductions during the nineteenth and twentieth centuries was mainly due to hunting, ecotourism, and the pet trade pathways (Biancolini *et al.*, 2021). Acclimatization societies were very active in South Africa and carried out numerous bird and mammal introductions to "improve" the aesthetic of the South-African landscape from a European point of view after the mid-1800s (B. W. van Wilgen *et al.*, 2020). In the last century, increasing global trade combined with the advent of the game-farming industry and ecotourism resulted in a striking rise in introductions of alien vertebrates and invertebrates (Picker & Griffiths, 2017; B. W. van Wilgen *et al.*, 2020).

As for other taxa, African regions with the earliest records of established alien species tend to have higher numbers of established alien species. For fishes, particularly high numbers of established alien species were recorded in North Africa due to Lessepsian invasion of marine species through the Suez Canal and to its closer socio-economic relationship with Europe **(Figure 2.30)**. Indeed, the number of alien fish in North Africa accelerated markedly after 1869 when the Suez Canal opened (Galil, 2000). In South Africa an increasing trend in established alien species detections is indicated as the number of marine alien species reported has increased from 15 (Griffiths *et al.*, 1992) to 95 established alien species (T. B. Robinson *et al.*, 2020). Although there is no doubt that new species are being introduced, other factors are also contributing to the increase in introductions, such as deeper historical analyses of past introductions (Mead *et al.*, 2011), varying levels of available taxonomic expertise across time (Griffiths *et al.*, 2009), and increased research efforts on underrepresented taxa or in under-studied ecosystems (T. B. Robinson *et al.*, 2020).

With the exception of plants, the introduction of alien species into freshwater systems in Africa has largely been

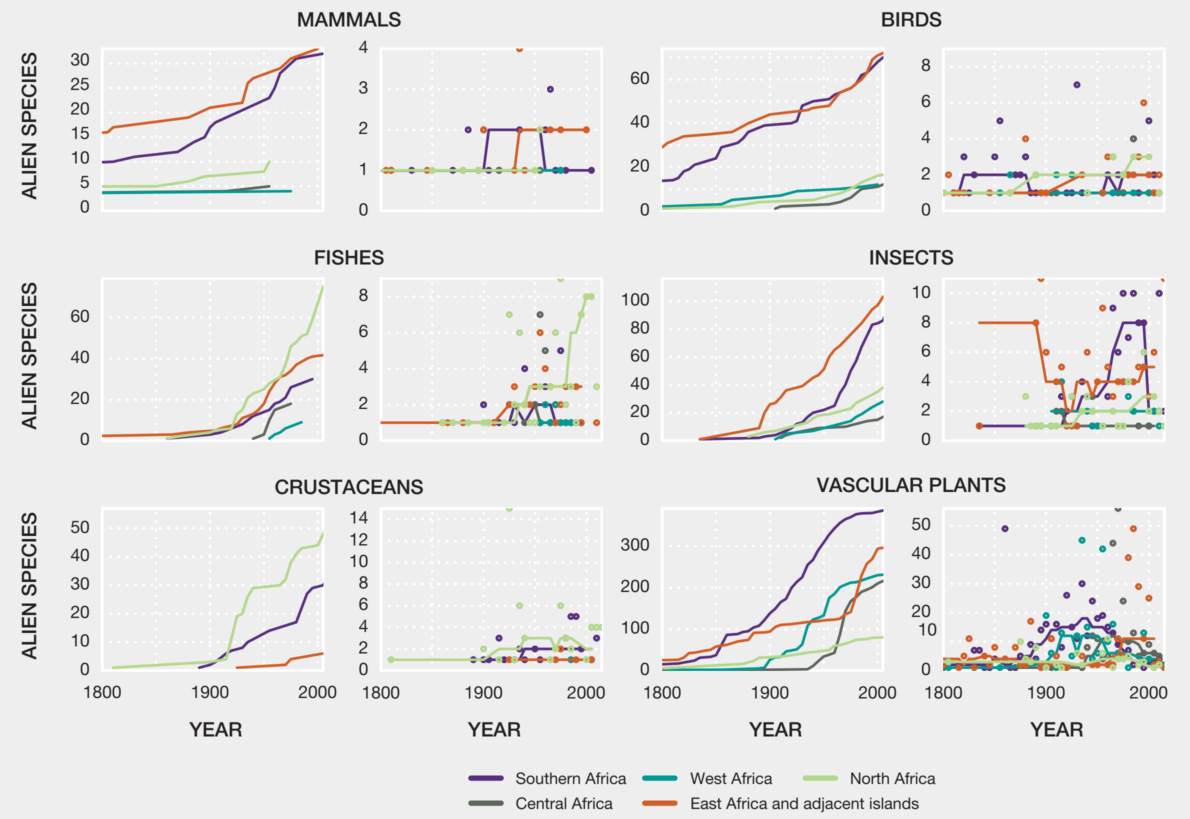

Figure 2 30 **Trends in numbers of established alien species for Africa.**

Panels show cumulative numbers (left panels) and numbers of established alien species per five-year intervals (right panels). Numbers here underestimate the actual extent of established alien species occurrences due to a lack of data. Lines in right panels indicate smoothed trends calculated as running medians (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582 intentional to enhance ecosystem services and promote nutritional, economic, or recreational values (Gherardi, Britton, *et al.*, 2011; Howard & Chege, 2007; Howard & Matindi, 2003; Munyaradzi & Mohamed-Katerere, 2006; Weyl *et al.*, 2020). However, the outcomes of these introductions were often opposite of the intended purpose, with losses of ecosystem function and services (B. W. van Wilgen *et al.*, 2020). For example, in South Africa the overall rate of alien freshwater animal introductions accelerated sharply after 1880 and generally increased over time, with unintentional introductions of invertebrates playing a relevant role (Weyl *et al.*, 2020). Only freshwater fish introductions underwent a significant decrease after the 1950s due to legislation regulating introductions and decreasing demand for new species for angling (Faulkner *et al.*, 2020). In general, the number of invertebrate introductions to South Africa rose over time (Faulkner *et al.*, 2016), this pattern being reported for freshwater (Weyl *et al.*, 2020), terrestrial (Janion-Scheepers & Griffiths, 2020), and marine invertebrate introductions (T. B. Robinson *et al.*, 2020).

Status

In light of Africa's colonial history, there have been surprisingly fewer introductions of alien mammals than to other regions (Long, 2003). Africa currently harbours 44 established alien mammals from seven orders and 18 families (Biancolini *et al.*, 2021). The most represented orders are Cetartiodactyla (17 species), Primates (9), Rodentia (7), and Carnivora (6). These alien species are mainly concentrated along the western Mediterranean coast, South Africa, and Madagascar and originate from within Africa (16), Europe and Central Asia (8), the Americas (8), and Asia and the Pacific (1). The pathways most frequently involved in alien mammal establishment were hunting (15 cases), the pet trade (10), farming (8), and conservation (8) (Biancolini *et al.*, 2021). Escaped game species are a growing problem in South Africa where numerous game-farming estates specialize in alien mammals (D. Spear & Chown, 2009; B. W. van Wilgen *et al.*, 2020). The status of these species is often classified as "within country" instead of alien as they are native to the geopolitical unit of South Africa. Nevertheless, they have been translocated outside of their historical native range (B. W. van Wilgen *et al.*, 2020). For example, *Tragelaphus angasii* (nyala), an antelope native to Africa, is now spreading outside its native range and possibly competing with native herbivores (Biancolini *et al.*, 2021; Downs & Coates, 2005). Of the 44 established alien mammal species, 27 (61.4 per cent) have ecological impacts (Biancolini *et al.*, 2021). For example, *Suncus murinus* (Asian house shrew), one of the "100 of the worst invasive alien species," has a patchy distribution from Madagascar to Egypt, and potentially has overlooked impacts on native plants, invertebrates, and small vertebrates through predation or competition (GISD, 2019). However, some alien mammal introductions were considered benign and carried out for conservation, such as for four primates threatened by habitat loss and translocated from their native mainland range to insular protected areas: *Daubentonia madagascariensis* (aye-aye), *Eulemur albifrons* (white-headed lemur), *Varecia variegata* (black-and-white ruffed lemur), and *Piliocolobus kirkii* (Zanzibar red colobus) (Andriaholinirina, Baden, Blanco, Chikhi, Cooke, *et al.*, 2014; Andriaholinirina, Baden, Blanco, Chikhi, Zaramody, *et al.*, 2014a, 2014b; Biancolini *et al.*, 2021; Davenport *et al.*, 2019).

Most alien bird species in Africa are found in the far south of the continent, although *Corvus splendens* (house crow) is distributed from Sudan to South Africa along the east coast. Most alien species are a legacy of Africa's European colonial past, such as *Fringilla coelebs* (chaffinch) and *Sturnus vulgaris* (common starling) in South Africa. Other notable alien birds in Africa are *Acridotheres tristis* (common myna) and *Passer domesticus* (house sparrow) (E. E. Dyer, Redding, *et al.*, 2017).

The number of alien reptile introductions in Southern Africa has risen in recent decades, but there is limited information about the trends elsewhere in this IPBES region (Capinha *et al.*, 2017; Kraus, 2009; Seebens, Blackburn, *et al.*, 2017; Van Wilgen *et al.*, 2010). For amphibians, many species have been translocated within Southern Africa (Measey *et al.*, 2017).

In contrast to most other taxa, the highest numbers of alien fishes and crustaceans – many marine – are found in North Africa **(Table 2.19)**. East Africa and its adjacent islands have the second highest numbers of alien fishes likely because of introductions in the many lakes of the Rift Valley area, including the three largest, Lakes Victoria, Tanganyika, and Malawi, that have high alien fish population densities and associated fisheries important for subsistence (Pitcher & Hart, 1995). In these lakes and large artificial reservoirs, *Lates niloticus* (Nile perch), *Limnothrissa miodon* (Tanganyika sardine), and tilapias are the main introduced fish species (Craig, 1992; Pitcher & Hart, 1995). Tilapias are tropical fishes in the family Cichlidae (mainly *Oreochromis, Tilapia*, and *Sarotherodon* spp.) that are native to parts of Africa and the Middle East but have been introduced globally mostly for aquaculture and human consumption (Canonico *et al.*, 2005). A total of 21 alien freshwater fishes have established in South Africa, and others have been translocated (Ellender & Weyl, 2014; Weyl *et al.*, 2020). The high number of alien fishes in Southern Africa is likely influenced by greater research efforts compared to other African regions. No alien marine fish have been reported for South Africa yet (T. B. Robinson *et al.*, 2020). Many freshwater fish have been intentionally introduced across Africa in order to maintain or increase fishery yields, enhance sport fisheries, or support the aquaculture industry (Darwall *et al.*, 2011; Ellender & Weyl, 2014; García *et al.*, 2010; Máiz-Tomé *et al.*, 2018).

By 2011, sixteen alien fish species had been introduced to Central Africa (Brooks *et al.*, 2011). In Madagascar, one quarter of the freshwater fish fauna consists of alien species, with 26 alien species present, of which at least 24 were deliberately introduced during the 1950s (Šimková *et al.*, 2019). On Île de la Réunion, six species of fish (and one decapod crustacean, *Macrobrachium rosenbergii* (giant freshwater prawn)) were introduced by 2002, but only four were established by then (Keith, 2002).

Notably, no review on introductions of freshwater alien species in Africa has been produced so far except for crayfish (Madzivanzira *et al.*, 2021). In other cases, current information is available only for specific taxa and has been only comprehensively and recently assessed for South Africa (M. P. Hill *et al.*, 2020; Weyl *et al.*, 2020; Zengeya & Wilson, 2020). Available data show that South Africa hosts 51 alien freshwater invertebrates and 32 alien freshwater fish, while 926 alien plant species are reported, and freshwater and terrestrial species are not distinguished (Zengeya & Wilson, 2020). Seventy-seven alien freshwater animals, largely dominated by fishes, molluscs, and crustaceans, are currently established in South Africa, most of which were intentionally introduced (Picker & Griffiths, 2017; Weyl *et al.*, 2020).

Among alien freshwater jellyfish, the cnidarian *Craspedacusta sowerbii* (peach blossom jellyfish) has been recorded in South Africa and potentially Morocco (Oualid *et al.*, 2019; Weyl *et al.*, 2020). Several species of alien molluscs have been recorded in African freshwaters, with 14 species of gastropods reported by 2011, some of which were released for the biological control of the intermediate hosts of schistosomiasis (Appleton, 2003; Appleton & Brackenbury, 1998). Only one alien freshwater bivalve *Corbicula fluminea* (Asian clam) has been recorded in African waters, an introduction probably related to fish stocking (Clavero *et al.*, 2012; Darwall *et al.*, 2011). Nine species of alien crayfish have been introduced to Africa, mostly for aquaculture. Five have established populations in the wild and three have spread widely in specific parts of Africa: *Procambarus clarkii* (red swamp crayfish) in Eastern Africa, *Cherax quadricarinatus* (redclaw crayfish) in Southern Africa, and *Procambarus virginalis* (Marmorkrebs) in Madagascar (Madzivanzira *et al.*, 2021).

Little is known about marine alien species in Africa. The most studied areas are along the South African coast which includes two large marine ecosystems, the Agulhas current in the east and the Benguela current in the west (Mead *et al.*, 2011; T. B. Robinson *et al.*, 2020). The total number of introduced marine species reported is 95, with 59 per cent considered as invasive alien species. A variety of taxa are represented, from the small protists (e.g., *Mirofolliculina limnoriae*) and dinoflagellates (e.g., *Alexandrium minutum*) to the most conspicuous macroalgae, molluscs, crustaceans, bryozoans, and tunicates. Most biological invasions were reported along the Benguela current large marine ecosystem (70 per cent) and alien species inhabit bays, estuaries, and artificial habitats, while only three are widespread and abundant on open rocky shores (the mussels *Mytilus galloprovincialis* (Mediterranean mussel) and *Semimytilus patagonicus*, and the barnacle *Balanus glandula*) (T. B. Robinson *et al.*, 2020). Angola harbours 29 introduced marine species, mostly concentrated in Luanda, the most studied area of the country (Pestana *et al.*, 2017). The most conspicuous and abundant taxa are bryozoans and tunicates, such as *Schizoporella errata* (branching bryozoan) and *Ascidiella aspersa* (European sea squirt), both global invasive alien species.

2.4.2.2 Plants

Trends

The number of established alien plant species in Africa has continually increased for centuries as reported for multiple African countries (Brundu & Camarda, 2013; L. Henderson, 2006; Maroyi, 2012; Senan *et al.*, 2012; Shaltout *et al.*, 2016). Southern Africa has experienced a steady increase in plant alien species numbers during the entire twentieth century, the most rapid rise of all African regions, and appeared to slow down only towards the end of the century **(Figure 2.30)**. In contrast, alien plant numbers in East Africa showed a marked acceleration starting in the final quarter of the twentieth century and have not yet slowed. In North Africa, alien plant numbers increased slowly but steadily towards the end of the nineteenth century. No readily apparent dynamics were detected for West Africa. However, this detected pattern is, to some extent, likely due to more intensive research and better data collected for the Republic of South Africa relative to the rest of the continent (Pyšek *et al.*, 2008; Pyšek, Pergl, van Kleunen, *et al.*, 2020).

Status

Southern Africa has the highest established alien species richness for all taxa (1,139) among all the subregions of Africa **(Table 2.19)**. Seven other countries harbour over 300 established alien plant species: Congo (522), Ethiopia (421), Morocco (410), Mozambique (396), Benin (333), Algeria (328), and Eswatini (315) (D. M. Richardson *et al.*, 2020). Expressed as the proportional contribution of established alien species to the national flora, countries that rank highest in this respect are Chad (12 per cent), Benin (11 per cent), and Eswatini (10 per cent); in South Africa, because of its extremely rich native flora, the contribution of established alien species to the total floristic richness of the country is only 5 per cent. South Africa also has the highest number of invasive alien species (374, D. M. Richardson *et al.*, 2020). *Bidens pilosa* (blackjack, occurring in 61 per cent of all African regions as defined by GloNAF corresponding mostly to countries), *Ricinus communis* (castor bean, 60 per cent), *Senna occidentalis* (coffee senna, 60 per cent), *Catharanthus roseus* (Madagascar periwinkle, 56 per cent), and *Euphorbia hirta* (garden spurge, 54 per cent) occur in more than half of the regions in Southern Africa. The following are the most widely distributed invasive alien plants in Southern Africa: *Lantana camara* (lantana, invasive in 46 per cent of regions), *Tithonia diversifolia* (Mexican sunflower), *Pontederia crassipes* (water hyacinth), *Chromolaena odorata* (Siam weed), *Leucaena leucocephala* (leucaena), *Prosopis juliflora* (mesquite, all invasive in more than 20 per cent of regions), and *Parthenium hysterophorus* (parthenium weed) (D. M. Richardson *et al.*, 2020). Concerning the donor regions of established alien plant species in Africa, the highest numbers were introduced from temperate Asia (19 per cent of all introductions to individual countries), Europe (13.9 per cent), tropical Asia (13.7 per cent), Southern America (13.4 per cent), and Northern America (10.9 per cent). However, 21 per cent of species that are established in African countries were introduced from another country on that same continent (van Kleunen *et al.*, 2015).

Alien tree species have had the greatest impact throughout Africa on biodiversity, water regimes, fire regimes, and ecosystem functioning (D. M. Richardson *et al.*, 2021). Many tree species used in forestry and agroforestry, especially *Eucalyptus* and *Pinus* (Pine), have been introduced throughout Africa, and some shrubs and trees such as *Acacia colei* (parta), *Acacia melanoxylon* (Australian blackwood), *Broussonetia papyrifera* (paper mulberry), *Calliandra houstoniana* (calliandra), *Calotropis gigantea* (yercum fibre), *Dahlia imperialis* (bell tree dahlia), *Ipomoea carnea* (pink morning glory), *Montanoa hibiscifolia* (tree daisy), and *Tecoma stans* (yellow bells) are well established in many parts of the continent (D. M. Richardson *et al.*, 2021). However, relative to *Pinus* and *Acacia*, *Eucalyptus* appears to have had a lower impact. South Africa's Mediterranean shrublands have been severely invaded by numerous alien trees and shrubs, especially species in the genera *Acacia*, *Hakea*, *Leptospermum* and *Pinus* (B. W. van Wilgen *et al.*, 2016). Australian *Acacia* species are actively promoted for agroforestry in other parts of the continent (D. M. Richardson *et al.*, 2004) and higher-lying areas have been heavily invaded by *Acacia melanoxylon* and *Acacia mearnsii* (black wattle), *Pinus patula* (Mexican weeping pine) and *Pinus radiata* (radiata pine). Pines and acacias are extremely invasive in the mountains of southwestern

Table 2 20 **Numbers of established alien species for subregions of Africa.**

For mammals, birds, and vascular plants ranges of values indicate variation among databases (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this table is available at https://doi.org/10.5281/zenodo.7615582

	Central Africa	East Africa and adjacent islands	North Africa	Southern Africa	West Africa	Total
Mammals	4-17	17-35	5-17	9-54	1-9	30-80
Birds	13-16	77-79	17-20	71-74	14-23	121-133
Fishes	26	56	130	46	17	187
Reptiles	2	33	8	124	9	158
Amphibians	0	5	2	2	5	12
Insects	33	143	71	227	48	344
Arachnids	9	29	10	70	11	94
Molluscs	2	11	75	67	7	142
Crustaceans	1	11	82	47	3	125
Vascular plants	880-1,071	1,738-2,570	485-1,162	1,754-2,292	645-818	3,109-4,498
Algae	3	4	42	12	1	58
Bryophytes	0	0	0	0	0	0
Fungi	19	44	18	82	9	122
Oomycetes	0	1	0	3	0	4
Bacteria and protozoans	1	2	1	2	1	4
Total	1,045-1,252	2,274-3,126	1,115-1,807	2,773-3,359	802-992	4,510-5,961

South Africa and in riparian habitats and other biomes (Holmes *et al.*, 2005). Other tree and shrub invaders with impacts include *Acacia dealbata* (acacia bernier), *Acacia decurrens* (green wattle), several *Rubus* (bramble) species, and *Biancaea decapetala* (Mysore thorn). *Azadirachta indica* (neem tree), *Prosopis juliflora* (mesquite), and *Leucaena leucocephala* (leucaena) are abundant invaders along the coastline of much of Africa, preferring hot and humid conditions. *Chromolaena odorata* (Siam weed) is now common in many countries in Central and Southern Africa, being abundant in open savanna grasslands, woodlands, riparian zones, forest gaps, and edges (D. M. Richardson *et al.*, 2021). **Table 2.20** lists the most widespread invasive alien species in Africa according to GRIIS.

Table 2 21 **Top most widespread invasive alien species for Africa.**

The number of regions where the respective species has been recorded and classified as being invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien species rather than their impacts, which is covered in **Chapter 4**. A maximum of three species is shown for each group (see **section 2.1.4** for further details about data sources and data processing). A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species name	No. of regions	Species name	No. of regions
Mammals		**Molluscs**	
Rattus rattus (black rat)	7	*Lissachatina fulica* (giant African land snail)	4
Mus musculus (house mouse)	6	*Pseudosuccinea columella* (mimic lymnaea)	3
Felis catus (cat)	5	*Bursatella leachii* (blue-spotted sea hare)	2
Birds		**Crustaceans**	
Corvus splendens (house crow)	9	*Penaeus monodon* (giant tiger prawn)	4
Acridotheres tristis (common myna)	4	*Cherax quadricarinatus* (redclaw crayfish)	3
Passer domesticus (house sparrow)	3	*Percnon gibbesi* (nimble spray crab)	2
Fishes		**Vascular plants**	
Poecilia reticulata (guppy)	9	*Lantana camara* (lantana)	31
Gambusia holbrooki (eastern mosquitofish)	7	*Pontederia crassipes* (water hyacinth)	30
Oreochromis niloticus (Nile tilapia)	6	*Chromolaena odorata* (Siam weed)	23
Reptiles		**Algae**	
Trachemys scripta elegans (red-eared slider)	3	*Caulerpa cylindracea* (green algae)	2
Hemidactylus frenatus (common house gecko)	2	*Alexandrium tamarense* (dinoflagellate)	1
Gehyra mutilata (mutilating gecko)	1	*Caulerpa chemnitzia* (green algae)	1
Amphibians		**Bryophytes**	
Rhinella marina (cane toad)	2		
Duttaphrynus melanostictus (Asian common toad)	1	**Fungi**	
Insects		*Ceratocystis fimbriata* (Ceratocystis blight)	1
Icerya purchasi (cottony cushion scale)	11	*Cryphonectria parasitica* (blight of chestnut)	1
Bactrocera cucurbitae (melon fly)	9	*Pseudocercospora fijiensis* (black Sigatoka)	1
Bactrocera dorsalis (Oriental fruit fly)	9	**Oomycetes**	
Arachnids			
Mononychellus tanajoa (cassava green mite)	1	**Bacteria and protozoans**	
Rhipicephalus microplus (cattle tick)	1	*Vibrio cholerae* (cholera)	9
		Yersinia pestis (black death)	1

By 2006, a total of 27 major invasive alien aquatic plants had been recorded in African waters, 16 alien to Africa, and 11 native to other parts of the continent (Howard & Chege, 2007). A recent review records the existence of 19 established alien freshwater plants only in South Africa, mainly introduced through trade and hitchhiking *via* boating and angling (M. P. Hill *et al.*, 2020). In South Africa, the most important invasive alien freshwater macrophyte remains *Pontederia crassipes* (water hyacinth), first recorded as established in KwaZulu-Natal in 1910. Four other species are also highly invasive, collectively referred to along with water hyacinth as the "Big Bad Five": *Pistia stratiotes* (water lettuce), *Salvinia × molesta* (kariba weed), *Myriophyllum aquaticum* (parrot's feather), and *Azolla filiculoides* (water fern) (M. P. Hill *et al.*, 2020; **Chapter 4**, **section 4.3.2.2**).

2.4.2.3 Microorganisms

In general, microbial biological invasions are more readily detected in well-surveyed regions, such as Europe, than in less well-surveyed regions, such as Africa, highlighting the importance of monitoring programmes at continental and inter-continental scale (Waage *et al.*, 2008). Fungi, oomycetes, and other microorganisms are poorly studied in most areas of the African continent. While Africa has been a source for several plant, animal, and human diseases (Bryant *et al.*, 2007; Costard *et al.*, 2009; Pretorius *et al.*, 2010), reports of biological invasions across most of Africa have declined over the years, except for South Africa (Zengeya *et al.*, 2020), most likely due to a lack of resources dedicated to this research. Thus, reliable data are scarce and mostly limited to a few well-researched regions, such as the Cape region (Crous *et al.*, 2006) where the introduction and impact of alien fungal species are best documented (Wood, 2017). In South Africa, nine alien pathogenic species are known to attack native plants, while 23 host-specific pathogens of alien plant species have likely been introduced together with their hosts (Wood, 2017). In addition, one fish pathogen, 11 alien saprotrophic species, and 61 species of alien fungi forming ectomycorrhizae have been reported (Wood, 2017). Furthermore, seven host-specific alien pathogens have been introduced for the biological control of invasive alien species (Wood, 2017).

Compared to other IPBES regions, Africa has the lowest number of known alien macrofungi, with 107 species (Monteiro *et al.*, 2020). Of these, 40 per cent belong to Agaricales, 29 per cent to Boletales and 13 per cent to Russulales. The most widespread macrofungi are *Pyrrhoderma noxium, Amanita muscaria* (fly agaric), *Pisolithus albu*s (white dye-ball fungus), *Rhizopogon luteolus* (yellow false truffle), and *Suillus granulatus* (weeping bolete mushroom), having been recorded for 8 or more countries. The highest numbers of alien macrofungi are reported for South Africa (65), Tanzania (25), Morocco (10), and Kenya (10). A number of countries, mainly from the Central African region, have between 1 to 5 known alien species.

2.4.2.4 Islands

Invasive alien species on islands are a major concern in the western Indian Ocean islands, including Comoros, Mauritius, Seychelles, Île de la Réunion, and smaller nearby islands where mammal predators such as cats and rats and plants negatively affect the increasingly disturbed ecosystems (Bonnaud *et al.*, 2011; Kueffer *et al.*, 2004; Russell *et al.*, 2016; Russell & Le Corre, 2009; Tassin & Laizé, 2015). Île de la Réunion is estimated to have over 2,000 alien plant species, with more than 100 of these classified as invasive (e.g., *Leucaena leucocephala* (leucaena), *Hiptage benghalensis* (hiptage), *Ulex europaeus* (gorse, Baret *et al.*, 2006; Soubeyran *et al.*, 2015). Of the 28 island groups, including 68 archipelagos present in the Western Indian Ocean, alien mammals can be found on each group with an average richness of five species per island group (Russell *et al.*, 2016). There are 12 invasive alien mammal species on Île de la Réunion and various combinations of six of them on the nearby Îles Éparses (Russell & Le Corre, 2009). The islands of East Africa are major hubs of alien reptiles and amphibians globally: Mauritius and Île de la Réunion are inhabited by 17 and 15 alien species, respectively (Capinha *et al.*, 2017; Kraus, 2009; Telford *et al.*, 2019). On Socotra, 88 alien plants have been recorded (Senan *et al.*, 2012). The recent invasion of Madagascar by *Duttaphrynus melanostictus* (Asian common toad) and some alien marine biota poses a severe threat to the native biodiversity of this island (Licata *et al.*, 2019; B. M. Marshall *et al.*, 2018). Similarly, the islands off the Western coast of Africa have repeatedly experienced animal invasions. In São Tomé and Principe, invasions began in the 1470s and by the end of the twentieth century, 14 alien mammal species were established on São Tomé and 12 on Principe (Dutton, 1994). Currently, 25 alien and invasive alien animal species are reported for both islands, of that 5 are birds, 2 ray-finned fish, 13 mammals, 4 insects, and 1 gastropod (De Menezes & Pagad, 2020). In Cabo Verde harbour there are 448 introduced plant taxa, equivalent to 60 per cent of the native flora, according to the Cabo Verde Biodiversity Database (Medina *et al.*, 2015). In addition, there are 38 alien and invasive alien animal species, including 4 ray-finned fishes, 2 gastropods and 2 marine invertebrates, 4 reptile species, 6 bird species, 10 mammal species, and 9 insect species (Martinez *et al.*, 2021).

2.4.2.5 Data and knowledge gaps

Although impacts of invasive alien species on Africa's biodiversity and ecosystem services are well known, there are still large gaps in scientific information (Egoh *et* *al.*, 2020; Faulkner *et al.*, 2015). With the exception of South Africa (B. W. van Wilgen *et al.*, 2020), these gaps are apparent in many subregions, particularly in East Africa and adjacent islands, both for units of analysis and many taxonomic groups. The number of documented alien species in many countries may be significantly underestimated as this is a function of information availability, research intensity, and country development status (McGeoch *et al.*, 2010).

For alien mammals, gaps exist for most of the African continent except for areas such as the western Mediterranean coast, South Africa, Madagascar, and adjacent islands. Knowledge of alien amphibians and reptiles is incomplete due to a lack of data (Capinha *et al.*, 2017; García-Díaz *et al.*, 2015; Kraus, 2009; Seebens, Blackburn, *et al.*, 2017; N. J. van Wilgen *et al.*, 2018). These gaps broadly match the distribution of data-deficient native reptile and amphibian species, which suggests a general scarcity of information about the status of reptiles and amphibians in the region (Böhm *et al.*, 2013; Stuart *et al.*, 2008). Further survey efforts in these data-poor areas can be expected to uncover established populations of alien amphibians and reptiles.

One of the main data gaps regarding freshwater invasions in Africa relates to the understanding of their geographical scope, given that most comprehensive reviews have been produced for South Africa only. A large taxonomic bias was also found, with reviews on faunal invasions, particularly fish invasions, or on specific species such as the highly invasive *Pontederia crassipes* (water hyacinth), dominating the literature, and many fewer studies on other taxonomic groups (Coetzee *et al.*, 2019). Thus, the status of alien and invasive alien species presented here certainly underestimates the true number of freshwater invasive alien species present in the region. Increased research could help to better inform the trends and status of freshwater invasive alien species in Africa.

For vascular plants, Africa is geographically covered completely by the GloNAF database (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015, 2019), providing data on alien plant species in individual countries, but of varying quality (Pyšek, Pergl, *et al.*, 2017) so that information remains scarce in some regions.

Information on the occurrence of alien fungi is missing for many African countries, mainly in North Africa, East Africa, and adjacent islands. The most complete information is available for South Africa, but even here knowledge is considered incomplete (Wood, 2017). The low number of alien macrofungi reported in most countries is likely a consequence of low research intensity and numbers are certainly underestimated.

2.4.3 Trends and status of alien and invasive alien species in the Americas

This section reports on the trends and status of alien species of the Americas **(Figure 2.31, Table 2.21)** for animals (**section 2.4.3.1**), plants (**section 2.4.3.2**), microorganisms (**section 2.4.3.3**), and islands (**section 2.4.3.4**), and provides an overview of data and knowledge gaps (**section 2.4.3.5**). A description of IPBES regions and sub-regions including a spatial representation is provided online (IPBES Technical Support Unit On Knowledge And Data, 2021) and in **Chapter 1**, **section 1.6.4**.

2.4.3.1 Animals

Trends

The number of alien animals in the Americas has increased across all taxonomic groups, especially post-1850, and across all subregions **(Figure 2.31)**. Particularly steep increases are observed for North America, followed by South America, with the exception of alien birds which also showed steep increases in the Caribbean. Since 1900 the rates of increase have remained stable (e.g., mammals), declining (fishes in North America), or distinctly increasing (arthropods). Increases in numbers of alien arthropods in North America have been shown in several studies (Aukema *et al.*, 2010; Mattson *et al.*, 1994; Nealis *et al.*, 2016) as well as in South America (Fuentes *et al.*, 2020), for freshwater (Ricciardi, 2001, 2006) and for marine animals (Carlton & Eldredge, 2009; Cohen & Carlton, 1998; Ruiz, Fofonoff, *et al.*, 2000). Transfers of species within a continent contribute to the spread and new incidences of alien species occurrences. Within the United States, for example, over 580 freshwater species have been introduced from one watershed to another outside their historical ranges; these introductions are nearly as numerous as those originating from outside the country, and they have increased over time, more than doubling in number since 1950 (USGS, 2021).

Alien mammal introductions in the Americas date to pre-Columbian times in the Caribbean islands for hunting (e.g., *Didelphis marsupialis* (common opossum), *Dasyprocta leporina* (agouti), *Dasypus novemcinctus* (nine-banded armadillo)) (Biancolini *et al.*, 2021; Giovas *et al.*, 2012; Long, 2003). European colonialism caused a surge in introductions of alien species beginning in the fifteenth century and peaking during the twentieth century, with a strong focus on game species and, more recently, on pets (Biancolini *et al.*, 2021; Long, 2003).Considered collectively, the number of alien amphibians and reptiles in the Americas has been increasing since the 1950s and the introduction of new alien species through the pet trade is predicted to either accelerate or remain steady (Kraus, 2009; Lockwood *et al.*, 2019; Perella & Behm, 2020; Powell *et*

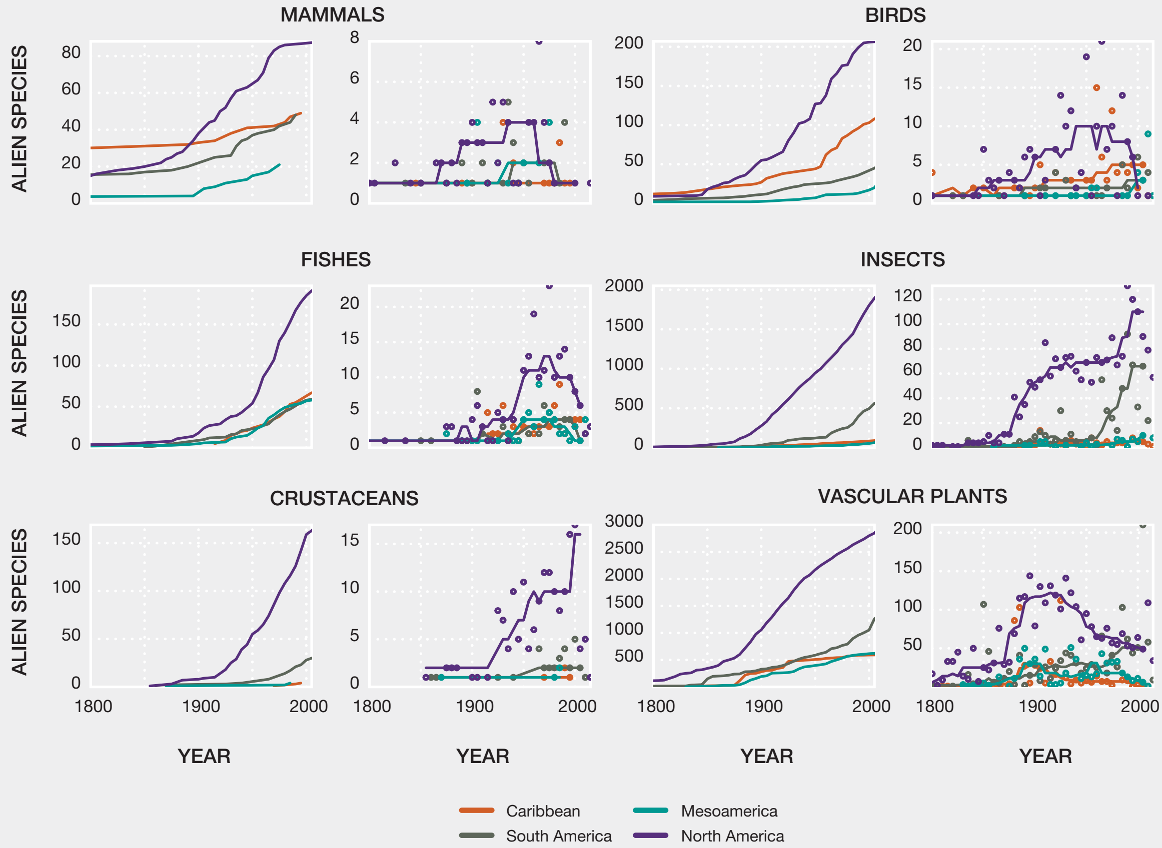

Figure 2 31 **Trends in numbers of established alien species for the Americas.**

The cumulative numbers (left panels) and number of established alien species per five-year intervals (right panels). Numbers shown here underestimate the real extent of alien species occurrences due to a lack of data. Lines in right panels indicate smoothed trends calculated as running medians (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

al., 2011; Seebens, Blackburn, *et al.*, 2017; Stringham & Lockwood, 2018).

The first introductions of alien aquatic species in South America occurred in the 1500s in conjunction with European colonization, but remained relatively low until the1800s and 1900s, when they moderately increased. Alien aquatic introductions began increasing distinctly in the mid 1900s, both in South and North America, as shown in **Figure 3.6** in the IPBES Regional Assessment Report on Biodiversity and Ecosystem Services for the Americas (IPBES, 2018b). Through the 2000s there has been a large increase in the number of records and studies of alien organisms (e.g., Frehse *et al.*, 2016; Vitule *et al.*, 2021). Current data trends show no signs of slowing, either in terms of the number of alien species or in new spatiotemporal records (e.g., Vitule *et al.*, 2021). Aquaculture and the aquarium trade (including e-commerce) are the most important pathways for the introduction of new alien species (e.g., Bezerra *et al.*, 2019; Magalhães *et al.*, 2020; Vitule *et al.*, 2019). Habitat alteration, the elimination of biogeographic barriers (e.g., D. A. dos Santos *et al.*, 2019; Vitule *et al.*, 2012), ballast water, hull fouling (Frehse *et al.*, 2016), and introducing fish for angling are other important mechanisms for introduction that have direct effects on both biodiversity and socio-economic aspects (e.g., Doria *et al.*, 2020; Vitule *et al.*, 2014).

For marine alien species in American waters, seminal studies have highlighted the rising numbers of marine alien species (Cohen & Carlton, 1998; Coles *et al.*, 1999). Recent updates for regions such as for the coastal waters of the American temperate zones found an increase in the total number of detected alien species, while the rate of newly recorded alien species has remained stable in recent decades (Bailey *et al.*, 2020). Teixeira & Creed (2020) reported that the number of introduced species increased by 160 per cent for Brazil between 2009 and 2019. A rise in the number of detected alien species was also found for

Argentina and Uruguay (Schwindt *et al.*, 2020), where the number of detections increased by a factor of 4.5 between 2001 and 2019, with an estimated arrival of one new species every 178 days.

Status

The Americas host a significant number of established alien mammals (96 species) from nine orders and 29 families. Most are from the orders Cetartiodactyla (30 species), Rodentia (28 species), Primates (14 species) and Carnivora (11 species) (Biancolini *et al.*, 2021). Within the Americas, alien mammal richness is high on the east coast of North America, Alaskan islands, Newfoundland Island, central-southern United States, the Caribbean Archipelago, and Patagonia (Malvinas) (Biancolini *et al.*, 2021). Many mammals native to the Americas have been translocated inside the region and are thus classified as being alien (53 species), while the major outside donors were Europe and Central Asia (8 species), followed by Asia and the Pacific (7 species) and Africa (2 species). Alien mammal introductions mainly occurred for sport hunting, the pet trade, so called "faunal improvement" (e.g., releases carried out to aesthetically modify the landscape), farming, and zoos (Biancolini *et al.*, 2021). A well-established hunting industry in North America fuels the introduction of ungulates, frequently contained in large enclosures in the southern United States and Mexico or directly released into the wild (Long, 2003). For example, *Ammotragus lervia* (aoudad), a bovid native to the Northern African savanna and desert areas, is now established in a large range north of Mexico (establishment not reported for Mexico) (Texas Invasive Species Institute, 2021). One of the most invasive alien mammals in the Americas is *Herpestes javanicus auropunctatus* (small Indian mongoose) established on many islands in the Caribbean (Biancolini *et al.*, 2021; Hays & Conant, 2007; Louppe *et al.*, 2020). This species was widely introduced during the nineteenth century as a biological control agent for rodents, and it is considered one of the "100 worst invasive alien species in the world" because of its generalist diet and high predatory efficiency. Another high-profile example of mammal invasion is the ongoing spread of *Hippopotamus amphibius* (so-called "Escobar's hippos"; hippopotamus) in the Magdalena River of Colombia (Biancolini *et al.*, 2021; Jarić *et al.*, 2020). Four individuals of this large African mammal were introduced by Pablo Escobar in the 1980s for his amusement and they escaped captivity in 1993 after his death (Dembitzer, 2017); in 2020, about 80–120 alien hippos were found to occur over 2000 km^2.

Alien bird species are particularly rich in North America, notably Florida and California, where several alien parrot species have established populations (E. E. Dyer, Cassey, *et al.*, 2017). Alien parrots are also widespread in South America. Attempts to establish all the bird species mentioned in Shakespeare's works into North America have a legacy in the distribution of *Sturnus vulgaris* (common starling) across the continents.

In South America, the number of reported alien aquatic organisms (ranging from microscopic fungi, invertebrates, and plants to large mammals (Schwindt *et al.*, 2018) is increasing rapidly (e.g., Fuentes *et al.*, 2020; Vitule *et al.*, 2021), with fish and molluscs (26.8 per cent and 25.2 per cent of studied invasive alien marine species respectively; see Schwindt & Bortolus, 2017, **Figure 2.31**) having the largest number of studies, species, and spatiotemporal occurrence records (e.g., (Bezerra *et al.*, 2019; Frehse *et al.*, 2016; Vitule *et al.*, 2021). The most recent records of fishes in South America indicate that over 75 alien species have been translocated between different basins within South America (Bezerra *et al.*, 2019; Vitule *et al.*, 2019) and more than 80 alien fish species have been introduced from other regions of the world (Doria *et al.*, 2021; Vitule *et al.*, 2019, 2021). Most of the alien aquatic species studied in South America belong to the salmonid and cichlid families, but *Limnoperna fortunei* (golden mussel) is the alien species included in the most publications within the region (Schwindt & Bortolus, 2017).

North America has a long and very well-studied history of aquatic species introductions, particularly for fish (e.g., Courtenay & Meffe, 1989; Fuller *et al.*, 1999; Moyle, 1986). Introductions of European and Asian species that have also been introduced worldwide are noteworthy, such as *Salmo trutta* (brown trout) or *Cyprinus carpio* (common carp), species of tropical or subtropical origin introduced to Florida, and species from elsewhere in the United States introduced to California, and more recently *Cyprinus carpio* in the Mississippi Basin. The Laurentian Great Lakes have many invasive alien animals of Ponto-Caspian origin **(Box 2.9)**, mostly introduced through ballast water (Ricciardi & MacIsaac, 2000; Vanderploeg *et al.*, 2002). *Pterois* species (lionfishes) have spread through the western Atlantic, including parts of North America and the Caribbean. The introduction of *Oreochromis niloticus* (Nile tilapia), *Salmo trutta, Cyprinus carpio*, and many other fish species is widespread throughout the Americas (e.g., Agostinho *et al.*, 2005; Contreras-Balderas *et al.*, 2008; Habit *et al.*, 2010, 2015). Similarly, many species native to small parts of the American continent (e.g., *Gambusia* spp. (Gambusias), *Oncorhynchus mykiss* (rainbow trout), *Poecilia reticulata* (guppy)) have been widely introduced throughout the Americas and elsewhere (Marr *et al.*, 2013).

The Americas is the IPBES region with the highest number of alien reptiles and amphibians **(Table 2.22)**. Within this region, the United States is home to several hotspots of alien amphibians and reptiles (Capinha *et al.*, 2017; Kraus, 2009; Krysko *et al.*, 2011, 2016). Florida (58 species established), California (25 species), and Puerto

Table 2 22 **Numbers of established alien species for subregions of the Americas.**

Numbers of alien species can vary depending on data sources. For mammals, birds and vascular plants, ranges of values indicate variation among databases (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this table is available at https://doi.org/10.5281/zenodo.7615582

	Caribbean	Mesoamerica	North America	South America	Total
Mammals	35-62	8-34	49-95	25-77	83-164
Birds	110-113	29-41	210-211	53-114	249-287
Fishes	91	226	619	144	803
Reptiles	60	60	121	56	192
Amphibians	20	8	41	16	62
Insects	153	163	2,116	640	2,636
Arachnids	33	36	168	76	207
Molluscs	26	60	212	68	255
Crustaceans	10	64	173	79	248
Vascular plants	1,402-1,761	1,600-2,242	6,571-7,424	2,492-3,099	8,005-9,325
Algae	4	105	65	50	193
Bryophytes	0	0	34	21	48
Fungi	17	15	174	219	363
Oomycetes	2	2	7	5	12
Bacteria and protozoans	1	4	6	5	14
Total	2,036-2,425	2,612-3,292	11,587-12,487	4,353-5,073	13,370-14,809

Rico (11 species) stand out as global hotspots of alien amphibians and reptiles (Capinha *et al.*, 2017; Kraus, 2009; Krysko *et al.*, 2011, 2016; Meshaka, 2011; Perella & Behm, 2020; Powell *et al.*, 2011). Besides Puerto Rico, other Caribbean islands such as Cuba and the Bahamas are also important global hotspots (Borroto-Páez *et al.*, 2015; Capinha *et al.*, 2017; C. R. Knapp *et al.*, 2011; Kraus, 2009; Powell *et al.*, 2011). In South America, Brazil is the country with the highest number of alien amphibians and reptiles, with a total of 136 species recorded, of which at least seven have established wild populations (Capinha *et al.*, 2017; É. Fonseca *et al.*, 2019; Kraus, 2009).

Marine alien species across the Americas are unequally studied geographically and taxonomically, and compilations are scarce over time and space. Comprehensive assessments are lacking even in well-studied regions, such as the United States, making it difficult to draw general conclusions (Bailey *et al.*, 2020). The first comprehensive assessment was made for the United States for continental coasts finding 298 marine alien species (Ruiz, Fofonoff, *et al.*, 2000). However, this assessment needs updating, that is, as of 2006 there are 257 introduced species in California alone (Ruiz *et al.*, 2011). The reports in the rest of North America and mesoamerica are spatially or taxonomically focused and no comprehensive compilations have been published. The Southwestern Atlantic is the best-known region in South America for marine invasive alien species, yet, unequally studied among countries and sub-regions (Schwindt & Bortolus, 2017). Brazil has the highest number of marine alien species with 138 species (Teixeira & Creed, 2020), followed by Argentina and Uruguay with 129 species (Schwindt *et al.*, 2020). On the Pacific coast, Chile reported 51 alien species (Castilla & Neill, 2009; Villaseñor-Parada *et al.*, 2017), and Colombia 4 (Gracia *et al.*, 2011), but this may be due to lack of research (Schwindt & Bortolus, 2017).

2.4.3.2 Plants

Trends

Over the last two centuries the cumulative rate of increase in established alien plant species was most rapid in North America, quickly accelerating at the end of the nineteenth century (**Figure 2.31**; Lavoie *et al.*, 2012; Pyšek *et al.*, 2019). South America exhibited a slower cumulative increase, likely due to fewer experts and lower research intensity when compared to North America (Frehse *et al.*, 2016; Schwindt *et al.*, 2020; Schwindt & Bortolus, 2017).(Fuentes *et al.*, 2008; Rojas-Sandoval & Acevedo-Rodríguez, 2015; Ugarte *et al.*, 2010). Numbers of alien plant species are expected to increase over the next

Table 2 23 **Top most widespread invasive alien species for the Americas.**

The number of regions where the species has been recorded and classified as being invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distributions of invasive alien species rather than their impacts which are covered in **Chapter 4**. A maximum of three species is shown for each group (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species name	No. of regions	Species name	No. of regions
Mammals		**Molluscs**	
Rattus rattus (black rat)	21	*Lissachatina fulica* (giant African land snail)	12
Mus musculus (house mouse)	19	*Melanoides tuberculata* (red-rimmed melania)	9
Rattus norvegicus (brown rat)	19	*Corbicula fluminea* (Asian clam)	8
Birds		**Crustaceans**	
Passer domesticus (house sparrow)	11	*Macrobrachium rosenbergii* (giant freshwater prawn)	6
Columba livia (pigeons)	10	*Cherax quadricarinatus* (redclaw crayfish)	5
Bubulcus ibis (cattle egret)	5	*Carcinus maenas* (European shore crab)	2
Fishes		**Vascular plants**	
Cyprinus carpio (common carp)	9	*Calotropis procera* (apple of sodom)	13
Oreochromis niloticus (Nile tilapia)	9	*Leucaena leucocephala* (leucaena)	13
Oncorhynchus mykiss (rainbow trout)	8	*Ricinus communis* (castor bean)	13
Reptiles		**Algae**	
Hemidactylus mabouia (tropical house gecko)	7	*Undaria pinnatifida* (Asian kelp)	4
Hemidactylus frenatus (common house gecko)	6	*Codium fragile* (dead man's fingers)	2
Anolis sagrei (brown anole)	4	*Didymosphenia geminata* (didymo)	2
Amphibians		**Bryophytes**	
Lithobates catesbeianus (American bullfrog)	11	*Campylopus introflexus* (heath star moss)	1
Rhinella marina (cane toad)	6	**Fungi**	
Xenopus laevis (African clawed frog)	4	*Batrachochytrium dendrobatidis* (chytrid fungus)	6
Insects		*Amanita phalloides* (death cap)	1
Icerya purchasi (cottony cushion scale)	11	*Bipolaris maydis* (southern corn leaf blight)	1
Maconellicoccus hirsutus (pink hibiscus mealybug)	11	**Oomycetes**	
Aedes albopictus (Asian tiger mosquito)	10	*Phytophthora cinnamomi* (Phytophthora dieback)	1
Arachnids		*Phytophthora lateralis* (Port-Orford-cedar root disease)	1
Raoiella indica (red palm mite)	7	*Phytophthora ramorum* (sudden oak death)	1
Aceria litchii (Litchi gall mite)	1	**Bacteria and protozoans**	
Avicularia avicularia (tarantula spiders)	1	*Vibrio cholerae* (cholera)	5
		Yersinia pestis (black death)	2

20 years in emerging South American economies such as Brazil, Mexico, and Argentina based on global trade dynamics and climate change (Seebens *et al.*, 2015) which could reverse the current status of North America as more invaded by plants than South America (Pyšek *et al.*, 2019).

Status

With 5,958 established alien vascular plant species, North America has the highest recorded alien plant richness in the world (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015). South America harbours 2,667 established alien plants (Pyšek *et al.*, 2019); note that the numbers differ from those presented in **Table 2.21**, because of different data sources and deviating data integration steps (**section 2.1.4** for further details). In the United States, California is the world's richest region in terms of established alien vascular plants with 1,753 established alien plant species, and Florida is a world regional hotspot with 1,473 established alien plants (Kartesz, 2014). *Sonchus oleraceus* (common sowthistle), *Plantago major* (broad-leaved plantain), *Taraxacum officinale* (dandelion), and *Poa annua* (annual meadowgrass) are among the most widely distributed established species in North America (each in more than 85 regions), while for South America the analogous list includes *Eleusine indica* (goose grass), *Sonchus oleraceus, Plantago major, Polygonum aviculare* (prostrate knotweed), and *Brassica rapa* (field mustard) (Pyšek, Pergl, *et al.*, 2017; **Table 2.23**). According to Pyšek, Pergl, *et al.* (2017), countries in Mesoamerica also harbour many established alien plants (Nicaragua 671, Mexico 519, Costa Rica 280, Panama 263), but due to their high native diversity, alien plants make up only 2.0–2.8 per cent of the total floras, the exception being Nicaragua with 10.4 per cent (e.g., Correa A. *et al.*, 2004; Pyšek, Pergl, *et al.*, 2017; Chacón & Saborío, 2012). Some regions in the Caribbean are heavily invaded by established alien plants, both in terms of actual species numbers (Cuba 542, Bahamas 356) or the proportion of established alien plants in the national floras (Bahamas 24 per cent, Barbados 14 per cent). Other countries in the Caribbean harbour 20 to 110 established alien plant species and their contributions to national floras do not exceed 8 per cent (Acevedo-Rodríguez & Strong, 2008; Kartesz, 2014; Pyšek, Pergl, *et al.*, 2017).

2.4.3.3 Microorganisms

Trends

The introduction of microorganisms has a long history in the Americas but is poorly documented as is the case worldwide. Where available, studies on the trends in alien microorganisms usually cover only fungi. For example, first records of alien fungi in Chile have been documented from the early twentieth century and show a continuous increase in numbers until the present (Fuentes *et al.*, 2020).

Status

The Americas harbour at least 199 alien macrofungi species, with approximately 36 per cent belonging to the group Agaricales, 32 per cent to Boletales and 11 per cent to Russulales (Monteiro *et al.*, 2020). Species most widely distributed within the region are *Suillus luteus* (ectomycorrhizal fungus of pine), *Amanita muscaria* (fly agaric), *Rhizopogon roseolus* (ectomycorrhizal fungus), and *Suillus granulatus* (weeping bolete mushroom). Countries with high numbers of known established species occur mainly in South America, and include Brazil (75), Argentina (60), and Chile (40) (Monteiro *et al.*, 2020). In the remaining IPBES sub-regions, higher numbers of known alien macrofungi were found in the United States (including Hawaii) (50), Canada, and Mexico (7 each).

2.4.3.4 Islands

Alien and invasive alien species are widespread on islands of both sides of the Americas: in the Pacific Ocean (notably the Galapagos islands) and the Atlantic Ocean (notably the Caribbean islands; e.g., (Kairo *et al.*, 2003; Rojas-Sandoval & Acevedo-Rodríguez, 2015; Van der Burg *et al.*, 2012). As an example, Caribbean Island forests are extensively dominated by alien tree species (Brandeis *et al.*, 2009; Chinea & Helmer, 2003; Helmer *et al.*, 2012), some of which are shade-tolerant and could permanently change forest species composition (C. J. Brown *et al.*, 2006). In addition, several alien species grow in forest plantations, livestock pastures, and abandoned agricultural fields creating both economic and environmental impacts. Such is the case for *Dichrostachys cinerea* (sickle bush), an alien species that occurs across almost 800,000 hectares in Cuba (Hernández *et al.*, 2002). The Hawaiian Islands are a global hotspot of plant invasions with 1,488 total alien plant species, and numbers for individual islands within the archipelago ranging from 386 to 913 alien species (Imada, 2012).

On the other side of the Americas, the Galapagos Archipelago harbours an estimated 1700 alien species with *Capra* sp. (goat) and *Rubus niveus* (Mysore raspberry) being among the most common until recently (Toral-Granda *et al.*, 2017). Between the 1980s and 1990s, the number of introduced plants has nearly doubled on the Galapagos Islands, reaching nearly 900 species (De Lourdes Torres & Mena, 2018). In addition, a study of the residence time and human-mediated propagule pressure of plants suggested that this archipelago is still in an early stage of plant invasions, due to the booming tourism industry and increasing human population size (Trueman *et al.*, 2010).

2.4.3.5 Data and knowledge gaps

Data availability for the Americas is dominated by studies from North America. Across taxonomic groups, the

Caribbean, Mesoamerica, and South America have considerably less data available relative to North America (Pyšek *et al.*, 2008). Studies on the temporal accumulation of alien species are almost exclusively available for this region except for a few studies for islands in the Caribbean and South America (Fuentes *et al.*, 2008; Rojas-Sandoval & Acevedo-Rodríguez, 2015; Toral-Granda *et al.*, 2017). Only a few studies on temporal trends exist for mainland South America or Mesoamerica (e.g., Fuentes *et al.*, 2020). Temporal information is scarce for most taxonomic groups in North America, including well-investigated groups such as vascular plants, birds, and mammals. For some groups, that are generally less studied globally, such as many invertebrates, fungi, and microorganisms, information is lacking for vast areas of this region.

In South America, regions often considered pristine and less impacted, such as the Amazon basin, lack studies on alien species and could be more thoroughly explored, particularly given recent levels of deforestation which could facilitate biological invasions (e.g., Frehse *et al.*, 2016; Vitule *et al.*, 2021; **Chapter 3**, **section 3.3.1**). In addition, there is a high degree of uncertainty on the status of alien species or populations and due to uncertainties about the native range of many species, the challenge of cryptic invasive alien species may be even greater for South America than the rest of the world (Bortolus *et al.*, 2015; Essl *et al.*, 2018; Jarić *et al.*, 2019).

A notable exception represents alien amphibians and reptiles which are relatively well-known in most of the Americas as a consequence of ongoing surveys and research (Capinha *et al.*, 2017; É. Fonseca *et al.*, 2019; García-Díaz *et al.*, 2015; González-Sánchez *et al.*, 2021; Kraus, 2009; Krysko *et al.*, 2016; Perella & Behm, 2020; N. J. van Wilgen *et al.*, 2018). Nevertheless, clarification of the status (i.e., being alien or native to a certain region) of some species in Mesoamerica and South America is needed (García-Díaz *et al.*, 2015; González-Sánchez *et al.*, 2021), and further work will improve the understanding of the ecology and impacts of the alien amphibians and reptiles present in this region (É. Fonseca *et al.*, 2019; N. J. van Wilgen *et al.*, 2018).

An important data gap exists for countries along the North Atlantic coast of South America (from French Guiana to Guiana; Schwindt & Bortolus, 2017). For example, in Venezuela the number of marine alien species originally reported by Pérez *et al.* (2007) was 22 but was later lowered to 11 alien species by Figueroa López and Brante (2020) due to uncertainty in the provided records. However, the number of marine alien species is likely higher even than the number reported by Pérez *et al.* (2007). No extensive compilations of alien species in general are available for continental Ecuador and for Peru (but see Calder *et al.*, 2021; Cárdenas-Calle *et al.*, 2019).

The availability of records on alien macrofungi for the Americas is dominated by a few countries, notably those for which higher numbers of alien species are reported here, including Argentina, Brazil, Chile and the United States. Important data gaps on established alien species exist for many other countries of the Americas, particularly in the Caribbean and Mesoamerica (Monteiro *et al.*, 2020). In general, information about alien microorganisms is lacking for all of the Americas as is the case for other IPBES regions.

2.4.4 Trends and status of alien and invasive alien species in Asia and the Pacific

This section reports on the trends and status of alien species of Asia and the Pacific for animals (**section 2.4.4.1**), plants (**section 2.4.4.2**), microorganisms (**section 2.4.4.3**), and islands (**section 2.4.4.4**), and provides an overview of data and knowledge gaps (**section 2.4.4.5**). A description of IPBES regions and sub-regions including a spatial representation is provided online (IPBES Technical Support Unit On Knowledge And Data, 2021) and in **Chapter 1**, **section 1.6.4**.

2.4.4.1 Animals

Trends

The numbers of alien animal species increased continuously for all taxonomic groups and all subregions of the Asia-Pacific regions **(Figure 2.32)**. The steepest increases were observed in Oceania for all animal groups considered in **Figure 2.32**, except for fishes. In Oceania, the number of alien animals rose distinctly already in the nineteenth century, much earlier relative to other subregions where steep increases were mostly observed after 1950. Northeast Asia experienced strong increases during that time for birds, fishes, and crustaceans. Likewise, increasing alien species numbers have been reported in various countries for insects (Huang *et al.*, 2011; Yamanaka *et al.*, 2015), gastropods (Barker, 1999; Roll *et al.*, 2009), amphibians and reptiles (Lee *et al.*, 2019), and marine alien species of different groups (Bailey *et al.*, 2020; Hewitt *et al.*, 2004).

Before colonization by Europeans, alien mammals in South-East Asia were introduced *via* ancient exchanges between the Indonesian Archipelago, Papua New Guinea, and Australia with numerous prehistoric introductions of game, fur, pet, and stowaway species (e.g., *Phalanger orientalis* (northern common cuscus), *Sus celebensis* (Sulawesi pig), *Dendrolagus matschiei* (Matschie's tree-kangaroo)) (Biancolini *et al.*, 2021; Heinsohn, 2003; Long, 2003). Introductions surged during the nineteenth century following

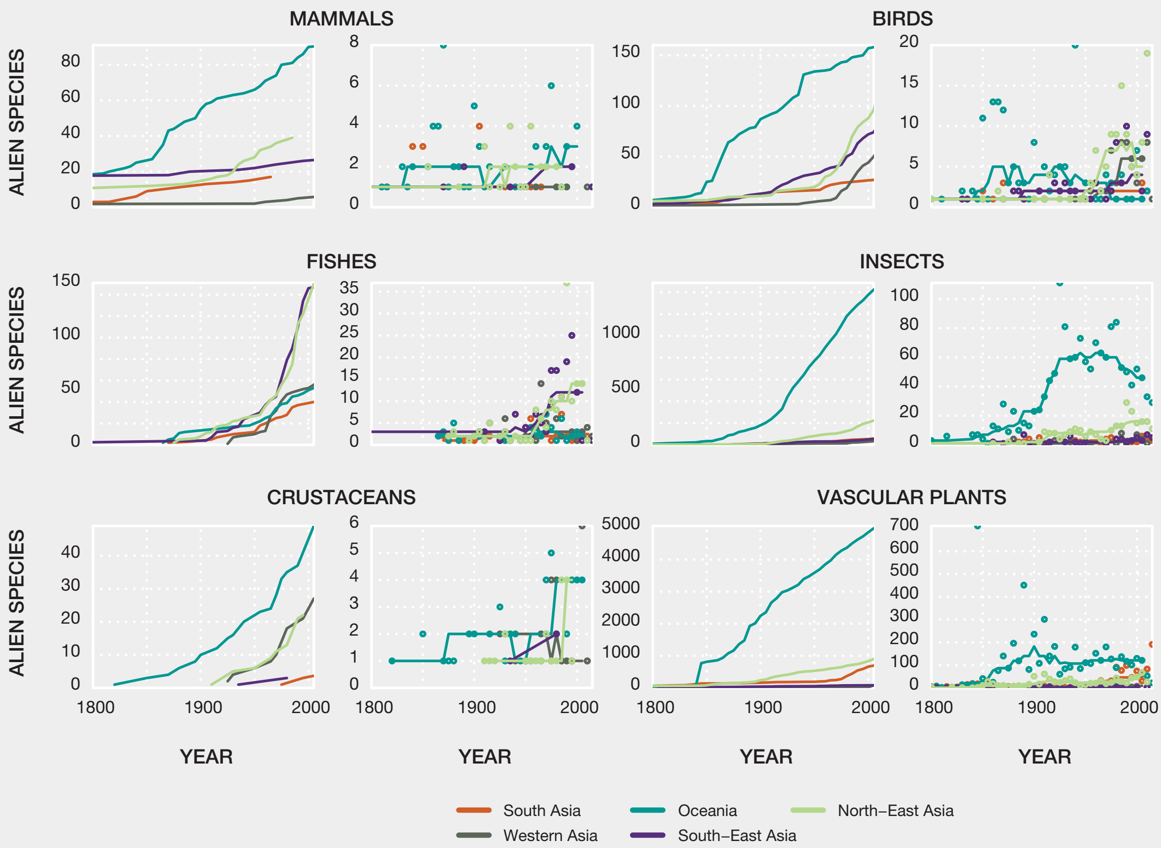

Figure 2 32 **Trends in numbers of established alien species for Asia and the Pacific.**

Cumulative numbers (left panels) and number of established alien species per five-year intervals (right panels). Numbers shown here underestimate the actual extent of alien species occurrences due to a lack of data. Lines in right panels indicate smoothed trends calculated as running medians (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

European colonization when Australia, New Zealand, and other Pacific islands became hotspots for alien mammals that negatively impacted native animal communities (Biancolini *et al.*, 2021; Woinarski *et al.*, 2015). The aim was to supply game species (e.g., *Cervus elaphus* (red deer), *Lepus europaeus* (European hare), *Dama dama* (fallow deer)) or create a familiar environment for colonists. In Central Asia and North-East Asia, alien mammal introductions were largely carried out at the beginning of the nineteenth century to create hunting and furbearing populations (Biancolini *et al.*, 2021; Clout & Russell, 2008; Long, 2003). Native Australian species became the subject of conservation introductions, also called assisted colonization, to offshore islands free of invasive alien mammals (Seddon *et al.*, 2015; Woinarski *et al.*, 2015).

The Asia-Pacific region has experienced a growing number of alien bird, reptile and amphibian introductions, a trend likely to continue in the future (Chapple *et al.*, 2016; Kraus, 2009; Lee *et al.*, 2019; Pili *et al.*, 2020; Seebens, Bacher, *et al.*, 2021; Seebens, Blackburn, *et al.*, 2017; Toomes *et al.*, 2020).

The number of alien freshwater species grew slowly in Asia and the Pacific until the nineteenth century **(Figure 2.32)** when the number of recorded alien freshwater species distinctly increased (H. H. Tan *et al.*, 2020; Yuma *et al.*, 1998). During the twentieth century, aquaculture was the main pathway for freshwater fish species introductions (Saba *et al.*, 2020; Xiong *et al.*, 2015) and in the beginning of the late twentieth century, many freshwater fish species were introduced for ornamental purposes (H. H. Tan *et al.*, 2020; Yuma *et al.*, 1998). The number of ornamental freshwater fish rapidly increased towards the end of the twentieth century and ornamental trade is now the main pathway of introduction (Goren & Ortal, 1999; Saba *et al.*, 2020; Yuma *et al.*, 1998).

As in many other regions, detected numbers of introduced alien marine species in the South Pacific region increased over time. The first assessment for New Zealand documented 129 alien species (Cranfield *et al.*, 1998), while the most recent assessment nearly doubled that number to 214 (Therriault *et al.*, 2021), with 15 alien species considered as new arrivals establishing between 2010 and 2018. Despite these numbers, recent work shows an apparent decline in primary detections since 2005 in several regions across Asia and the Pacific. It is unknown if this decline is a result of effective preventive strategies (**Chapter 5**, **section 5.5.1**) or a reduction in search effort (Bailey *et al.*, 2020). In Asia, alien species introductions occur mainly by unintentional translocations such as ballast water discharged in ports located across China's coast (Y. Chen *et al.*, 2017).

Status

Asia and the Pacific is the region with the highest number of established alien mammals in the world (130 species), from 12 orders and 34 families (Biancolini *et al.*, 2021). The majority are from the orders Cetartiodactyla (30), Diprotodontia (28), Rodentia (26) and Carnivora (21). Areas with high numbers of alien mammals are Japan, the Indonesian archipelago, Australia, New Zealand, and the Pacific islands. These alien species originate mainly from within the region itself (96), while 14 alien species originate from Europe and Central Asia, 13 from the Americas, and 10 from Africa. Major pathways of alien mammal introductions in Asia and the Pacific are hunting (48 alien species), conservation (28), pet trade (27), faunal improvement (27), farming (22), stowaway transportation (16), and biocontrol (12) (Biancolini *et al.*, 2021). During the nineteenth century acclimatization societies sought to "improve" local fauna by introducing many aesthetically pleasing and/or game species to Australia and New Zealand (Biancolini *et al.*, 2021; Simberloff & Rejmanek, 2011). Of the 130 established alien mammal species, 68 (52 per cent) have invasive alien populations (Biancolini *et al.*, 2021). Examples include the prolific generalist *Oryctolagus cuniculus* (rabbits), a well known invasive alien species in Australia (Kirkpatric *et al.*, 2008), and the generalist *Trichosurus vulpecula* (brushtail possum), which was introduced to New Zealand in 1858 for domestic fur and meat trade (Forsyth *et al.*, 2018; Gormley *et al.*, 2012).

Despite Asia and the Pacific having a larger area and more suitable habitats than Europe and Central Asia, the Asia-Pacific region harbours similar numbers of alien amphibians and reptiles as Europe and Central Asia **(Table 2.18)**. This pattern may possibly be a result of stringent biosecurity measures (**Chapter 5**, **section 5.6.3.3**) in some areas such as Australia, New Zealand, and Japan, (Brenton-Rule *et al.*, 2016; Chapple *et al.*, 2016; García-Díaz *et al.*, 2017; Toomes *et al.*, 2020), but also lower relative research intensity in other regions **(Figure 2.6)**. Despite the comparatively low alien species richness, the Asia-Pacific region harbours two of the best-known examples of alien reptiles and amphibians, namely *Boiga irregularis* (brown tree snake) in Guam and *Rhinella marina* (cane toad) in Australia and other Pacific islands (Engeman *et al.*, 2018; Lever, 2003; Rogers *et al.*, 2017; Shine, 2018; Zug, 2013). The notable invasive alien species *Lithobates catesbeianus* (American bullfrog), *Trachemys scripta* (pond slider), and *Eleutherodactylus planirostris* (greenhouse frog) have been reported in China (S. Lin *et al.*, 2017; X. Liu *et al.*, 2015; Shi *et al.*, 2009). Additionally, Japan (17 alien species), the Cook Islands (14 alien species), and island territories such as Taiwan, Province of China, (at least 12 alien species) and Guam, United States (11 alien species) are global hotspots of alien amphibians and reptiles (Capinha *et al.*, 2017; Kraus, 2009; Lee *et al.*, 2019; Zug, 2013).

In Asia, the number of introduced alien freshwater species is highest for China (439) (Xiong *et al.*, 2015), followed by Malaysia (203 freshwater fishes) (Saba *et al.*, 2020) and the Philippines (159 freshwater fishes) (Casal *et al.*, 2007). The number of established alien freshwater fishes is highest in China (61) (Luo *et al.*, 2019), followed by Singapore (42) (H. H. Tan *et al.*, 2020), the Philippines (39) (Casal *et al.*, 2007), and Japan (23) (Yuma *et al.*, 1998). Most of the established alien fishes were introduced for aquaculture (Casal *et al.*, 2007; Luo *et al.*, 2019), while the proportion of introduced ornamental fishes is much lower (Casal *et al.*, 2007; Luo *et al.*, 2019).

A regional assessment of marine alien species across Asia and the Pacific is lacking, and, as in many other marine regions, records are likely underestimated. Lutaenko *et al.* (2013) compiled an atlas of marine invasive alien species in the Northwest Pacific Region, which includes territories from Japan, the Republic of Korea, the Russian Federation and China (Yellow Sea). For Japan, 42 marine alien species were reported (Iwasaki, 2006), mostly concentrated in eutrophicated enclosed bays near large urban cities such as Tokyo Bay and Osaka Bay. Although ballast water and hull fouling are important vectors, 21 species were reported as intentionally introduced for commercial sales, live bait, or fishery studies (Lutaenko *et al.*, 2013). Partial updates were done by Doi *et al.* (2011) adding crustaceans (mainly crabs, amphipods, barnacles, and isopods) to the list of alien species reported by Iwasaki (2006), increasing the 42 by 10 reported alien species. There are few reports about marine species introductions to Korean and Chinese waters. Seo and Lee (2009) reported 136 species suspected to be alien across this vast region of Asia, while 41 alien species are recognized only for the Republic of Korea (Lutaenko *et al.*, 2013). As for the Russian waters of the Northwest Pacific region, 37 marine invasive alien species were reported by 2010 and this number increased to 66 in a later assessment (Zvyagintsev *et al.*, 2011), mostly concentrated around Peter the Great Bay in Russia. Two recent reports for the north Pacific document 73 alien species for the northern central

Table 2 24 **Numbers of established alien species for subregions of Asia and the Pacific.**

Numbers of established alien species can vary depending on data sources. For mammals, birds, and vascular plants ranges of values indicate variation among databases (see **section 2.1.4** for further details on data sources and data processing). Note numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this table is available at https://doi.org/10.5281/zenodo.7615582

	North-East Asia	Oceania	South Asia	South-East Asia	Western Asia	Total
Mammals	28-53	50-105	12-28	38-54	5-20	97-163
Birds	119-129	169-175	29-38	84-85	84-139	287-336
Fishes	287	95	90	296	125	633
Reptiles	41	41	7	35	13	103
Amphibians	24	13	4	12	1	43
Insects	607	1,521	111	89	101	2,017
Arachnids	67	83	13	18	6	129
Molluscs	81	119	15	24	89	261
Crustaceans	43	75	12	19	63	158
Vascular plants	2,219-2,454	4,631-6,747	1,055-3,142	1,313-1,598	271-562	6,141-9,101
Algae	55	63	8	13	47	157
Bryophytes	0	32	0	0	0	32
Fungi	59	303	17	20	1	363
Oomycetes	9	5	2	1	0	12
Bacteria and protozoans	7	4	3	2	4	12
Total	4,008-4,278	7,963-10,140	1,490-3,602	2,053-2,355	932-1,293	10,445-13,520

Indo-Pacific, 208 species for the northwest Pacific (includes northeast Asia), and 368 for the northeast Pacific (from the United States, Canada up to Alaska; Kestrup *et al.*, 2015; Lee II & Reusser, 2012). In conclusion, the vast region of the north Pacific has a similar number of introduced marine species as the Mediterranean Sea. In addition, the northwest Pacific contains the largest number of alien fishes (34 species), most intentionally released into the wild or maintained in aquaculture facilities.

There are few exhaustive assessments for the south Pacific Ocean with the greatest research efforts in Australia and New Zealand. Surveysof Port Phillip Bay (Australia) detected 100 marine alien species (Hewitt *et al.*, 2004). A subsequent thorough literature reviewthat included data from port surveys yielded 132 alien species (Sliwa *et al.*, 2008). As of March 2018 in New Zealand, 214 established alien species were reported (Therriault *et al.*, 2018). The knowledge of marine bioinvasions of the Pacific Island Countries and Territories is scattered and dispersed in diverse publications. Surveys in Pago Pago Harbor (American Samoa) recognized 17 marine alien species (Coles *et al.*, 2003), 40 alien species were detected from Guam (Paulay *et al.*, 2002), and 11 alien species in Malakal harbour, Palau (M. L. Campbell *et al.*, 2016). Most alien species were associated with transport in ballast water or biofouling (Hewitt & Campbell, 2010), and the number of intentional introductions for aquaculture purposes are low in Australia and New Zealand but high across the Pacific Islands countries (Eldredge, 1994). Many introduction attempts have been conducted in the past 50 years in the south Pacific Ocean, with at least 38 alien species originating from small scale fisheries or aquaculture activities.

2.4.4.2 Plants

Trends

First records of alien plant species in Asia and the Pacific date back more than 1000 years (Wijesundara, 2010), and continual increases in the number of established alien species have been consistently recorded for several Asian and Pacific countries (Banerjee, 2020; C. Chen *et al.*, 2017; Jaryan *et al.*, 2013; Lazkov & Sultanova, 2011; Shrestha, 2016; Vinogradov & Kupriyanov, 2016; Wijesundara, 2010; Wu *et al.*, 2010). The most dramatic increase in the cumulative number of alien plant species is recorded for Oceania, including Australia, New Zealand, and the Pacific Islands **(Figure 2.32)**. Introduction rates peaked in around 1900, followed by a decline and a re-acceleration in the mid-twentieth century **(Figure 2.32)**. The trends for other Asia-Pacific sub-regions are similar to that for Oceania but they have markedly lower absolute numbers of established alien species per time period.

Table 2 25 **Top most widespread invasive alien species for Asia and the Pacific.**

The number of regions where the species has been recorded and classified as being invasive based on GRIIS (Pagad *et al.*, 2022). Note this table only refers to the distribution of invasive alien species rather than their impacts which are covered in **Chapter 4**. A maximum of three species is shown for each group (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species name	No. of regions	Species name	No. of regions
Mammals		**Molluscs**	
Rattus rattus (black rat)	23	*Lissachatina fulica* (giant African land snail)	15
Mus musculus (house mouse)	18	*Pomacea canaliculata* (golden apple snail)	11
Rattus norvegicus (brown rat)	14	*Euglandina rosea* (rosy predator snail)	6
Birds		**Crustaceans**	
Acridotheres tristis (common myna)	16	*Amphibalanus improvisus* (bay barnacle)	3
Columba livia (pigeons)	7	*Cherax quadricarinatus* (redclaw crayfish)	3
Corvus splendens (house crow)	7	*Procambarus clarkii* (red swamp crayfish)	3
Fishes		**Vascular plants**	
Gambusia holbrooki (eastern mosquitofish)	16	*Lantana camara* (lantana)	29
Cyprinus carpio (common carp)	15	*Pontederia crassipes* (water hyacinth)	28
Gambusia affinis (western mosquitofish)	12	*Leucaena leucocephala* (leucaena)	23
Reptiles		**Algae**	
Hemidactylus frenatus (common house gecko)	4	*Alexandrium minutum* (dinoflagellate)	2
Iguana iguana (iguana)	4	*Caulerpa taxifolia* (killer algae)	1
Trachemys scripta elegans (red-eared slider)	4	*Chattonella marina* (raphidophyte)	1
Amphibians		**Bryophytes**	
Lithobates catesbeianus (American bullfrog)	6		
Rhinella marina (cane toad)	6	**Fungi**	
Xenopus laevis (African clawed frog)	2	*Pyrrhoderma noxium*	4
Insects		*Amanita muscaria* (fly agaric)	1
Solenopsis geminata (tropical fire ant)	14	*Austropuccinia psidii* (myrtle rust)	1
Tapinoma melanocephalum (ghost ant)	14	**Oomycetes**	
Brontispa longissima (coconut hispine beetle)	13	*Phytophthora cinnamomi* (Phytophthora dieback)	3
Arachnids		**Bacteria and protozoans**	
Aculops lycopersici (Tomato russet mite)	1	*Vibrio cholerae* (cholera)	3
Latrodectus geometricus (brown widow spider)	1	*Yersinia pestis* (black death)	1
Latrodectus hasselti (Redback spider)	1		

Status

The Asia-Pacific region includes several global hotspots of established alien plant species (Dawson *et al.*, 2017) as for islands in Oceania (Essl *et al.*, 2019; Moser *et al.*, 2018). Such hotspots include New Zealand with 1,726 established alien plant species (comprising 44.5 per cent of the flora; Howell & Sawyer, 2006), Tahiti with 1,346 (73.8 per cent), and Guam with 833 (66.5 per cent, Raulerson, 2006). Australian states harbour from 1,186 established alien species in Western Australia to 1,584 in New South Wales, corresponding to 12–25 per cent of the total plant diversity in these states (Pyšek, Pergl, *et al.*, 2017; Randall, 2002; Walsh & Stajsic, 2007). Australasia experienced a rapid accumulation of established alien plants during colonization, while the Pacific islands show the steepest increase in established plant species among all global regions (van Kleunen *et al.*, 2015). The most widespread established alien species on the Pacific Islands include *Euphorbia hirta* (garden spurge), *Cenchrus echinatus* (southern sandbur), *Phyllanthus amarus* (jamaicaweed), *Sida rhombifolia* (arrowleaf sida), *Carica papaya* (papaya), *Eleusine indica* (goose grass), and *Euphorbia prostrata* (prostrate sandmat). In Australia and New Zealand,the most widespread established alien species are *Sonchus oleraceus* (common sowthistle), *Solanum americanum* (American black nightshade), *Chenopodiastum murale* (nettle-leaf goosefoot), *Medicago polymorpha* (bur clover), and *Malva parviflora* (pink cheeseweed) (Pyšek, Pergl, *et al.*, 2017; **Table 2.25**). Global hotspots of established alien species also occur in other Asian sub-regions; in South Asia and South-East Asia, India (471 alien plants comprise 2.6 per cent of the flora; Inderjit *et al.*, 2018), the Philippines (628 species, 6.4 per cent; Pelser *et al.*, 2011), and Indonesia (503 species, 1.7 per cent; Biotrop, 2003) are invasion hotspots. In Nepal, 21 established alien plant species have been classified as being invasive (Shrestha, 2016), while 101 invasive alien plant species have been recorded for Bhutan (Dorjee *et al.*, 2020). In North-East Asia, Japan is richest in alien plants (1311 species, 22.6 per cent) and numbers from China range from 100 to 400 (Pyšek, Pergl, *et al.*, 2017). Western Asia is comparatively poor in numbers of alien plants (**Table 2.25**; Pyšek, Pergl, *et al.*, 2017).

2.4.4.3 Microorganisms

Trends

In general, information on the trends of alien microorganisms in Asia is very scarce as for other IPBES regions. Data from China indicate that of the 27 invasive alien fungi recorded so far, only two new additions were reported after the year 2000 (H. G. Xu & Qiang, 2018). In India, only one new invasive alien fungal pathogen (*Puccinia horiana* (white rust of chrysanthemum)) has been recorded in the last five years (Akhtar *et al.*, 2019; Dubey *et al.*, 2021). However, 15 invasive fungal pathogens were intercepted by plant quarantine (Akhtar *et al.*, 2019, 2021; Dubey *et al.*, 2021) between 2015 and 2020. Only scattered information on trends of invasive alien fungi is available from other countries in Asia.

Status

Twenty-seven invasive alien fungal pathogens were recorded from China (H. G. Xu & Qiang, 2018), 21 from India (Akhtar *et al.*, 2019, 2021; Dubey *et al.*, 2021; Government of India, 2005), 30 from the Maldives (Shafia & Saleem, 2003), and 15 from the Lao People's Democratic Republic (Nhoybouakong & Khamphouke, 2003). Further information on invasive alien fungi is not traceable or available from countries in Asia though it is clear from studies by Fisher *et al.* (2020) that several new invasive alien fungi may have been introduced from across the globe.

A comparatively high number of known alien macrofungi has been reported for Asia and the Pacific which harbours at least 235 established alien species (Monteiro *et al.*, 2020). Most of these alien species belong to the order Agaricales (54 per cent), followed by Boletales (21 per cent), and Russulales (10 per cent). The most widespread alien macrofungi is *Pyrrhoderma noxium*. The countries with the highest numbers of known alien macrofungi are New Zealand (170 species) and Australia (40 species). This highlights the paucity of knowledge on invasive alien microparasites in this region. In general, it is assumed that goods, species including humans constantly carry a multitude of microorganisms around the globe and that many of them are introduced every year without detection.

2.4.4.4 Islands

Many islands in the Asia-Pacific region are significantly impacted by invasive alien species (IPBES, 2018b). For example, French Polynesia has undergone severe invasions by species ranging from avian malaria, plants, mammals, ants, birds, and predatory land snails (Brodie *et al.*, 2014; Howarth, 1985; J.-Y. Meyer, 2014; J.-Y. Meyer & Butaud, 2009). Mammals are widely introduced on islands in Asia and the Pacific (Courchamp *et al.*, 2003), with examples including commensal rodents (mice, black rats, brown rats, and Pacific rats), rabbits, pigs, goats, cats, and foxes, in particular on many islands (D. J. Campbell & Atkinson, 2002; Priddel *et al.*, 2000; Reaser *et al.*, 2007; St Clair, 2011; Towns *et al.*, 2006).

Conversely, while some islands are invaded by only a few alien species, they are archetypal examples of island invasions. Invasive *Herpestes* sp. (mongooses) have been introduced on the Japanese islands of Amami-Oshima and Okinawa (Goldson, 2011; Reaser *et al.*, 2007; The Ministry of the Environment of Japan, 2014). On Guam, *Boiga irregularis* (brown tree snake) has spread widely reaching densities in excess of 31,000 individuals per km^2 (CGAPS, 1997; Fritts & Rodda, 1998; Rogers *et al.*, 2017). On Guam and on Christmas Island, *Anoplolepis gracilipes* (yellow crazy ant) invasions were boosted by the invasive *Tachardiaephagus tachardiae* (yellow lac scale insect), which supplies yellow ants with honeydew (O'Dowd *et al.*, 2003; Reaser *et al.*, 2007). Other typical examples are gastropod invasions on many Polynesian islands, such as *Lissachatina fulica* (giant African land snail; Tsatsia & Jackson, 2022). Invasive plants are also a serious issue on many Asia-Pacific islands, such as Tahiti (J.-Y. Meyer & Florence, 1996), Lord Howe Island (T. D. Auld & Hutton, 2004), and Carnac Island (Abbott *et al.*, 2000), while invasive soilborne plant pathogens, such as the fungus *Phytophthora cinnamomi* (Phytophthora dieback), are problematic in over 70 countries including several Australian islands (T. D. Auld & Hutton, 2004; Pickering *et al.*, 2007), Fiji, Samoa, Tuvalu, and New Zealand (e.g., F. Campbell, 2010; Thaman, 2011; Thaman & O'Brien, 2011). Hawaii is another classic example of an archipelago heavily invaded by many species groups, being among the three regions with the most established alien species in the world (Dawson *et al.*, 2017): over 1,000 plants (W. L. Wagner *et al.*, 1999), 3,000 arthropods (Nishida, 2002), and 110 vertebrates (Moulton & Pimm, 1983; Vitousek *et al.*, 1987).

Pacific Islands are widely invaded by alien birds with New Zealand being a global hotspot of alien bird richness. More than 130 species were introduced to New Zealand, mostly deliberately by acclimatization societies set up by British colonists. More than 30 species are now established, including dense populations of several passerine species imported from Britain, such as *Turdus philomelos* (song thrush), *Turdus merula* (Eurasian blackbird), *Prunella modularis* (dunnock), *Chloris* sp. (greenfinch), *Acanthis* sp. (redpoll) and *Emberiza citrinella* (yellowhammer).

2.4.4.5 Knowledge and data gaps

For alien plants, Asia and the Pacific have lower data coverage than other continents; data are available on established alien species for 68.5 per cent of the area of tropical Asia as a whole (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015). Notable exceptions represent some well-studied invasion hotspots such as Australia, New Zealand and Hawaii (van Kleunen *et al.*, 2015, 2019; **Figure 2.6**). Mainland Asia is a region especially affected by knowledge gaps for alien mammals, likely due to a low sampling effort (Pyšek *et al.*, 2008) and/or linguistic barriers (Angulo *et al.*, 2021). Notably, while reports of alien mammals in Hong Kong, Special Administrative Region of China, are numerous and exhaustive, very little information is available in English for mainland China (Biancolini *et al.*, 2021). However, the situation has improved recently with several specialized accounts published or underway (Dorjee *et al.*, 2020; Inderjit *et al.*, 2018; Patzelt *et al.*, 2022), and this trend is expected to continue. Temporal information such as first records is generally scarce for most regions in Asia and the Pacific.

The completeness of the information about alien amphibians and reptiles and freshwater species in Asia and the Pacific varies substantially by country. While some countries in North-East Asia and Oceania are relatively well-studied, others, particularly in southeast Asia and western Asia, have substantial knowledge gaps (Capinha *et al.*, 2017; Chapple *et al.*, 2016; C. Chen *et al.*, 2017; Cogălniceanu *et al.*, 2014; Das, 2015; Kraus, 2009; Lee *et al.*, 2019; Rights and Resources Initiative, 2015; Seebens, Blackburn, *et al.*, 2017; Soorae *et al.*, 2010; Van Wilgen *et al.*, 2010; Zug, 2013). In addition, further genetic work is needed to resolve the status of various species and populations of alien reptiles throughout the Pacific and western Asia (Cogălniceanu *et al.*, 2014; Zug, 2013).

The total number of marine alien species varies among studies, in part due to a lack of standardized terminology, sampling methods, environments studied, and taxonomic expertise available to assess species lists and record dates (Marchini *et al.*, 2015). For example, many species counted as marine alien species in the northwest Pacific are present in aquaculture facilities, while it remains unknown whether they have established in some cases. Some assessment lists only include species detected on vectors, some others consider different delineations of marine regions, while yet others are country specific.

Asia and the Pacific is grossly under-explored for alien fungi and other microorganisms. The high number of alien macrofungal records in New Zealand and Australia are likely influenced by high research and sampling intensities in these regions. Much less data and fewer studies on alien macrofungi are available for most other countries in Asia and the Pacific.

2.4.5 Trends and status of alien and invasive alien species in Europe and Central Asia

This section reports on the trends and status of alien species of Europe and Central Asia for animals (**section 2.4.5.1**), plants (**section 2.4.5.2**), microorganisms (**section 2.4.5.3**) and islands (**section 2.4.5.4**), and provides an overview of data and knowledge gaps (**section 2.4.5.5**). A description of IPBES regions and sub-regions including a spatial representation is provided online (IPBES Technical Support Unit On Knowledge And Data, 2021) and in **Chapter 1**, **section 1.6.4**.

2.4.5.1 Animals

Trends

The number of alien animal species in Europe and Central Asia has increased across various taxonomic groups including vertebrates (Rabitsch & Nehring, 2017), insects (Roques *et al.*, 2016), molluscs (Peltanová *et al.*, 2012) and freshwater species (Muñoz-Mas & García-Berthou, 2020; Nunes *et al.*, 2015). Comparisons of long-term trends among sub-regions show much larger numbers of alien species recorded for Central and Western Europe, which has the highest numbers of alien species for all animal groups and at all times, compared to other sub-regions **(Figure 2.33)**. While rates of increase remained relatively constant over the last 200 years for alien mammals, they rose sharply in recent decades for birds and invertebrates. Rates of increase of alien species remained relatively constant for all groups in Eastern Europe, but available numbers in Central Asia are often too low to assess trends **(Figure 2.33)**.

Alien mammal introductions first occurred in Europe and Central Asia during prehistoric times, with major introductions from Asia to Europe and from the mainland to islands (Biancolini *et al.*, 2021; Long, 2003). The spread of agriculture brought domestic species (e.g., *Capra hircus* (goats), *Ovis aries* (sheep), *Felis catus* (cat)), while island colonization by humans brought game species (e.g., *Lepus europaeus* (European hare), *Glis glis* (European edible dormouse), *Oryctolagus cuniculus* (rabbits)) as well as stowaways (*Apodemus sylvaticus* (long-tailed field mouse), *Crocidura suaveolens* (lesser white-toothed shrew), *Microtus arvalis* (common vole)) (Biancolini *et al.*, 2021; Long, 2003). Biological invasions of islands intensified with the growth of the sea trade in the following centuries causing the disappearance of many natural island ecosystems, especially in the Mediterranean basin (Masseti, 2009). Hunting has always been and continues to be a major pathway for alien mammals and birds on both the mainland and the islands of Europe and Central Asia (Genovesi *et al.*, 2012; Monaco *et al.*, 2016).

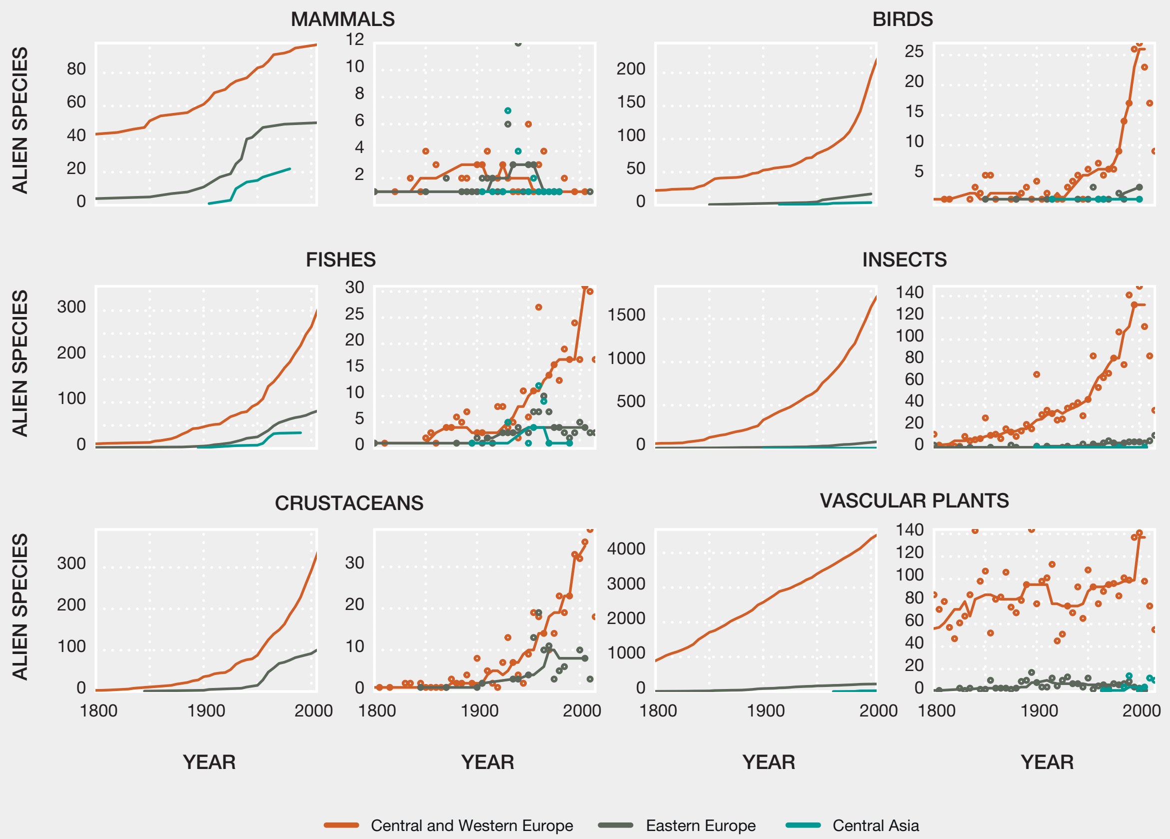

Figure 2.33 **Trends in numbers of established alien species in Europe and Central Asia.**

Cumulative numbers (left panels) and number of established alien species per five-year intervals (right panels). Numbers underestimate the actual extent of alien species occurrences due to a lack of data. Lines in right panels indicate smoothed trends calculated as running medians (**section 2.1.4** for further details about data sources and data processing). Note that presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

The number of alien animal species in Europe and Central Asia has increased across various taxonomic groups including vertebrates (Rabitsch & Nehring, 2017), insects (Roques *et al.*, 2016), molluscs (Peltanová *et al.*, 2012) and freshwater species (Muñoz-Mas & García-Berthou, 2020; Nunes *et al.*, 2015). Comparisons of long-term trends among sub-regions show much larger numbers of established alien species recorded for Central and Western Europe, the highest species numbers for all animal groups and at all times, compared to other sub-regions **(Figure 2.33)**. While rates of increase remained relatively constant over the last 200 years for alien mammals, they rose sharply in recent decades for birds and invertebrates. Rates of increase of established alien species remained relatively constant for all groups in Eastern Europe, but available numbers in Central Asia are often too low to assess trends **(Figure 2.33)**.

Alien mammal introductions first occurred in Europe and Central Asia during prehistoric times, with major introductions from Asia to Europe and from the mainland to islands (Biancolini *et al.*, 2021; Long, 2003). The spread of agriculture brought domestic species (e.g., *Capra hircus* (goats), *Ovis aries* (sheep), *Felis catus* (cat)), while island colonization by humans brought game species (e.g., *Lepus europaeus* (European hare), *Glis glis* (European edible dormouse), *Oryctolagus cuniculus* (rabbits)) as well as stowaways (*Apodemus sylvaticus* (long-tailed field mouse), *Crocidura suaveolens* (lesser white-toothed shrew), *Microtus arvalis* (common vole)) (Biancolini *et al.*, 2021; Long, 2003). Biological invasions of islands intensified with the growth of the sea trade in the following centuries causing the disappearance of many natural island ecosystems, especially in the Mediterranean basin (Masseti, 2009). Hunting has always been and continues to be a major pathway for alien mammals and birds on both the mainland and the islands of Europe and Central Asia (Genovesi *et al.*, 2012; Monaco *et al.*, 2016).

Europe and Central Asia has experienced a growing number of alien reptile and amphibian introductions, a trend that will likely continue (Seebens, Bacher, *et al.*, 2021; Seebens, Blackburn, *et al.*, 2017). Trends in alien reptiles and amphibians follow a similar pattern: historical events and trade routes around the Mediterranean Basin have resulted in some of the oldest known introductions of alien amphibians and reptiles in the world occurring in this region (Mateo *et al.*, 2011; Pleguezuelos, 2002). In line with global trends, the number of alien amphibians and reptiles has increased in this region and the pet trade is expected to contribute more species in the near and medium futures (Capinha *et al.*, 2017; Filz *et al.*, 2018; Kraus, 2009; Mateo *et al.*, 2011).

Introductions of alien freshwater animals increased after the mid-nineteenth century due to the activities of acclimatization societies, mainly for angling (Gherardi *et al.*, 2009). Established alien species numbers also increased notably after World War II due to more intensive trade, openings of major inland canals and waterways in Central and Western Europe, and the intensification of aquaculture (Gherardi *et al.*, 2009; Nunes *et al.*, 2015). The main pathways of introduction were releases and escapes through aquaculture, deliberate stocking, and pet and aquarium trades. The latter acquired more importance as a driver facilitating introductions since the 1990s (Nunes *et al.*, 2015). In central and north-eastern Europe, interconnected canals and waterways were the main pathways of introduction, while in Central and Western Europe releases and escapes are linked to aquaculture and pet and aquarium trades. A slight decrease in introduction rates in recent decades has been reported on the Iberian Peninsula (Muñoz-Mas & García-Berthou, 2020). Alien species introductions are further assisted by unintentional translocations, such as the opening of waterways in Israel (Goren & Ortal, 1999).

Across the coastal areas of Europe, the number of detections and introductions of alien species has increased over time, although numbers differ among assessments (Bailey *et al.*, 2020; Gollasch, 2006; Katsanevakis *et al.*, 2020; Tsiamis *et al.*, 2019), especially for the eastern Mediterranean Sea since the earliest inventories taken during the 1960s (Galil *et al.*, 2021b). For example, the number of marine alien species along the coast of Israel has increased three-fold from 1970 (147 alien species) to 2020 (452 alien species), and this trend is consistent as new alien species detections still appear in the scientific literature. For the Baltic Sea, the annual introduction rate has more than doubled since 1950: 1.4 species per year between 1950 and 1999 and 3.2 between 2000 and 2018 (ICES, 2019).

Status

Currently 85 alien mammals are known to be established in Europe and Central Asia, from 7 orders and 24 families (Biancolini *et al.*, 2021). The most numerous orders are Rodentia (26 species), Cetartiodactyla (24), Carnivora (18) and Eulipotyphla (8). Alien mammal hotspots are present in Central and Western Europe, numerous Mediterranean islands, the British Isles, Italy, Scandinavia, Eastern Europe and European Russia (Biancolini *et al.*, 2021). Most alien mammals are native to other parts of Europe and Central Asia (42) and the major outside donor is Asia and the Pacific (14), followed by the Americas (10), and Africa (4). This great reshuffling of mammal fauna was mainly driven by hunting (36 cases), pet trade (22), stowaway transportation (16), intentional introductions (12), conservation purposes (11) and fur exploitation (11) (Biancolini *et al.*, 2021). For example, squirrels were released or escaped from captivity in the last several decades, creating numerous alien populations scattered across Europe (Biancolini *et al.*,

Table 2 26 **Numbers of established alien species for subregions of Europe and Central Asia.**

For mammals, birds, and vascular plants ranges of values indicate variation among databases (**section 2.1.4** for further details about data sources and data processing). Note presented numbers may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this table is available at https://doi.org/10.5281/zenodo.7615582

	Central and Western Europe	Central Asia	Eastern Europe	Total
Mammals	64-133	5-23	24-80	72-164
Birds	218-627	4-5	20-24	221-630
Fishes	423	51	119	469
Reptiles	94	0	6	98
Amphibians	42	2	5	43
Insects	2,698	28	213	2,747
Arachnids	289	2	6	289
Molluscs	557	4	75	584
Crustaceans	420	10	88	563
Vascular plants	4,498-7,896	134-361	1,950-2,400	5,146-8,519
Algae	483	0	82	526
Bryophytes	23	0	1	23
Fungi	594	3	28	609
Oomycetes	59	0	2	59
Bacteria and protozoans	22	0	2	23
Total	12,711-16,587	265-511	2,903-3,413	11,472-15,346

2021). A well-known example is *Sciurus carolinensis* (grey squirrel), which was introduced to the United Kingdom and Italy (Bertolino *et al.*, 2008, 2014; Gaertner *et al.*, 2016). *Ondatra zibethicus* (muskrat), *Nyctereutes procyonoides* (raccoon dog), and *Mustela vison* (American mink) are among the most widespread species in Europe and Central Asia (Biancolini *et al.*, 2021; Genovesi *et al.*, 2012; Tedeschi *et al.*, 2022).

Many alien bird species were introduced during European colonial expansion including a large number introduced to Europe. Game and ornamental species were particularly popular, such that Europe now has populations of a number of alien galliforms and wildfowl. Other such introductions pre-date colonialism, such as *Phasianus colchicus* (common pheasant), which is widespread in Europe and still released in various countries every year by the tens of millions. Prior to the bird flu epidemic of 2005, Europe was a major hub for the caged bird trade, but European Union-wide bans on imports have greatly restricted the influx of species from outside the continent (Reino *et al.*, 2017). There is still extensive trade in captive-bred birds within Europe, and escapes continue to threaten further alien species introductions. The caged bird trade is the major source of alien species in Asia, notably in trade hubs in the Far East. Millions of birds continue to be trapped from wild populations in Asia, and pose a substantial extinction threat to popular species, as well as a risk of new alien populations.

Europe and Central Asia have several global hotspots of alien amphibians and reptiles. These include the Balearic Islands (20 species), mainland Spain (13 species), mainland Italy (11 species), mainland France (10 species), and the United Kingdom (10 species) (Capinha *et al.*, 2017; Ficetola *et al.*, 2010; Kark *et al.*, 2009; Kraus, 2009; Mateo *et al.*, 2011). Fewer alien reptiles and amphibians have been reported from Central Asian countries than in Europe (Capinha *et al.*, 2017; Kraus, 2009).

According to Nunes *et al.* (2015), there are 534 alien freshwater animals (46 per cent native to some European areas) in Europe and Central Asia. The Iberian Peninsula, France, Italy, the United Kingdom, and Germany host the highest numbers of species (Gollasch & Nehring, 2006; R. P. Keller *et al.*, 2009; Muñoz-Mas & García-Berthou, 2020; Nunes *et al.*, 2015; Teletchea & Beisel, 2018). For Uzbekistan, 31 alien freshwater fishes have been recorded (Yuldashov, 2018). Most introduced fish arrived mainly through stocking, aquaculture, or pet and aquarium trades, followed by crustaceans and molluscs, both mainly *via* ornamental trade and through corridors (e.g., canals and waterways; Muñoz-Mas & García-Berthou, 2020; Nunes *et al.*, 2015). Some species, such as *Cyprinus carpio* (common carp), *Sander lucioperca* (pike-perch), *Silurus glanis* (wels catfish) or Ponto-Caspian gobies, are only native to parts of western Europe but have now established in much of European fresh waters (e.g., Leprieur *et al.*, 2008). Similarly, many widespread species such as *Perca fluviatilis* (perch), *Rutilus rutilus* (roach) or *Alburnus alburnus* (bleak) are not native to the peninsulas in southern Europe, which have distinct, threatened fish faunas with high endemism (Yuldashov, 2018).

2.4.5.2 Plants

Trends

Since the start of the nineteenth century, Central and Western Europe has had a steady increase in alien plant introductions and data indicate no deceleration of this trend **(Figure 2.33)**. First records for Eastern Europe and Central Asia show very slow increases, partly due to lower research effort in these regions relative to Central and Western Europe (**section 2.4.5.5**). A recent Europe-wide inventory of established alien plants, including Central and Western, and a portion of Eastern Europe was conducted through the project Delivering Alien Invasive Species In Europe (Lambdon *et al.*, 2008) and recorded 4,139 established alien plant taxa (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015), an increase of 390 taxa (or 9.6 per cent). The introduction of alien aquatic plants increased after 1950, the main pathway being the ornamental trade, followed by cultivation and contaminants of commodities (Nunes *et al.*, 2015). Ornamental trade and cultivation had similar rates in different European areas while contaminants of commodities were mostly recorded in southern Europe (Nunes *et al.*, 2015). The number of alien aquatic plant species is still relatively low in European freshwaters but is sharply increasing, having doubled in nearly 30 years (Hussner *et al.*, 2010).

Status

In Central and Western Europe, a total of 8,565 alien vascular plants, 497 established alien algae, and 25 established alien bryophytes have been recorded **(Table 2.27)**. The GloNAF database reports 4139 established alien vascular plants (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015). The highest numbers of established alien plants are recorded in England (1,379), Sweden (874), Scotland (861), Wales (835), France (716), Norway (595), Belgium (508), Italy (478), Spain (454), and Germany (451) indicating that the northern part of the continent, particularly United Kingdom, Ireland, and Scandinavia are heavily invaded by established alien species. Only a few regions in Eastern Europe (perhaps due to lack of data) harbour comparably high numbers of established alien species, such as the European part of Russia (649), Ukraine (626) and Bulgaria (593) (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2015, 2019). Some of these countries also have the highest per centage of established alien species as a proportion of the total flora. In England, established alien species make up 47 per cent of the total flora, in Wales 44 per cent, Scotland 42 per cent, Sweden 35 per cent, in Norway 32 per cent, and in the European part of Russia 37 per cent (Pyšek, Pergl, Dawson, *et al.*, 2020). There are 35 alien species that have become established in more than 30 regions of Europe, that is, at least half of the European regions considered in the GloNAF database, the most widespread being *Erigeron canadensis* (Canadian fleabane; recorded in 76 per cent of regions), *Elodea canadensis* (Canadian pondweed), *Matricaria discoidea* (rounded chamomile), *Oenothera biennis* (common evening primrose), *Solidago canadensis* (Canadian goldenrod) and *Galinsoga parviflora* (gallant soldier) **(Table 2.27)**. Central Asia is generally less invaded by alien plants with country floras in this region harbouring 50–70 established alien species which corresponds to 1.9–4.5 per cent contribution to total plant diversity (Pyšek, Pergl, *et al.*, 2017).

According to Nunes *et al.* (2015), there are 210 alien freshwater plants (38 per cent native to some European areas). Hussner (2012) found that the highest number of alien plant species in all of Europe is reported for Italy and France, followed by Germany, Belgium, Hungary, and the Kingdom of the Netherlands. The most frequently introduced plants are the angiosperms: 200 out of 210 (Nunes *et al.*, 2015).

Over last decade, negative impacts associated with the spread of particular alien aquatic plant species (e.g., *Elodea* spp. (waterweeds), *Pontederia crassipes* (water hyacinth), *Ludwigia* spp. (primrose-willow), *Hydrocotyle ranunculoides* (floating pennywort), *Myriophyllum aquaticum* (parrot's feather)) increased in Europe (Hussner, 2012). Even though the number of alien aquatic plants appears relatively small compared to alien terrestrial plant species, the European and Mediterranean Plant Protection Organization (EPPO, 2021) has listed 18 of these species as invasive or potentially invasive within the European and Mediterranean Plant Protection Organization's region covering most of Europe and parts of Central Asia and North Africa. In total, 96 aquatic alien species from 30 families have been

Table 2 27 **Top most widespread invasive alien species for Europe and Central Asia.**

The number of regions where the species has been recorded and classified as invasive based on GRIIS (Pagad *et al.*, 2022). Note this table refers only to the distribution of invasive alien species rather than their impacts which are covered in **Chapter 4**. A maximum of three species is shown for each group (see **section 2.1.4** for further details about data sources and data processing). "No. of regions" denotes the number of regions with confirmed occurrences of that species according to the chapter database. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

Species name	No. of regions	Species name	No. of regions
Mammals		**Molluscs**	
Mustela vison (American mink)	15	*Dreissena polymorpha* (zebra mussel)	15
Rattus norvegicus (brown rat)	11	*Corbicula fluminea* (Asian clam)	13
Myocastor coypus (coypu)	10	*Potamopyrgus antipodarum* (New Zealand mudsnail)	13
Birds		**Crustaceans**	
Alopochen aegyptiaca (Egyptian goose)	8	*Pacifastacus leniusculus* (American signal crayfish)	18
Branta canadensis (Canada goose)	7	*Amphibalanus improvisus* (bay barnacle)	14
Psittacula krameri (rose-ringed parakeet)	6	*Faxonius limosus* (Spiny-cheek crayfish)	14
Fishes		**Vascular plants**	
Pseudorasbora parva (topmouth gudgeon)	19	*Ailanthus altissima* (tree-of-heaven)	32
Lepomis gibbosus (pumpkinseed)	18	*Robinia pseudoacacia* (black locust)	31
Gambusia holbrooki (eastern mosquitofish)	15	*Solidago canadensis* (Canadian goldenrod)	26
Reptiles		**Algae**	
Trachemys scripta (pond slider)	6	*Sargassum muticum* (wire weed)	7
Trachemys scripta elegans (red-eared slider)	4	*Coscinodiscus wailesii* (diatom)	5
Chelydra serpentina (common snapping turtle)	2	*Bonnemaisonia hamifera* (red algae)	4
Amphibians		**Bryophytes**	
Lithobates catesbeianus (American bullfrog)	7	*Campylopus introflexus* (heath star moss)	10
Pelophylax ridibundus (Eurasian marsh frog)	3	*Orthodontium lineare* (cape thread-moss)	2
Triturus carnifex (Italian crested newt)	3	**Fungi**	
Insects		*Ophiostoma novo-ulmi* (Dutch elm disease)	9
Cameraria ohridella (horsechestnut leafminer)	13	*Hymenoscyphus fraxineus* (ash dieback)	5
Harmonia axyridis (harlequin ladybird)	12	*Ophiostoma ulmi* (Dutch elm disease)	4
Leptinotarsa decemlineata (Colorado potato beetle)	8	**Oomycetes**	
Arachnids		*Aphanomyces astaci* (crayfish plague)	13
Opilio canestrinii (harvestman)	3	*Phytophthora cambivora* (root rot of forest trees)	3
Varroa destructor (Varroa mite)	3	*Phytophthora ramorum* (sudden oak death)	3
Mermessus trilobatus (trilobate dwarf weaver)	2	**Bacteria and protozoans**	
		Anabaenopsis raciborskii (cyanobacteria)	1
		Erwinia amylovora (fireblight)	1

reported as established alien species from at least one European country. Sixteen alien species belong to the family of Hydrocharitaceae, followed by the Nymphaeaceae and Lemnaceae (both with nine plant species). Most aquatic alien plant species introduced into Europe are native to North America (26 per cent) and Asia (29 per cent) (Hussner, 2012). The highest number of aquatic alien plant species was found in Italy (34 species), France (34 species), Germany (27), Belgium, and Hungary (both 26), and was lowest in the Balkan region and the northern and eastern parts of Europe (Hussner, 2012). *Elodea canadensis* (Canadian pondweed) is the most widely distributed alien aquatic plant in Europe, occurring in 41 European countries (but not in Cyprus, Malta, Iceland, Greece, and Montenegro). *Azolla filiculoides* (water fern) is the second most widely distributed species (25 countries), followed by *Vallisneria spiralis* (eelweed) (22) and *Elodea nuttallii* (Nuttall's waterweed) (20) (Hussner, 2012).

2.4.5.3 Microorganisms

Trends

Due to global trade of live plants and animals, the rate of introduction of alien fungi, oomycetes, and other microorganisms to Europe and Central Asia is likely to further accelerate (Hulme, 2021). Several fungi, oomycetes, and other microorganisms causing diseases have been introduced in recent decades (Nunes *et al.*, 2015). For example, within the past 20 years, 5 downy mildew pathogens with the potential to cause significant losses have been introduced to Europe (Gilardi *et al.*, 2013; Görg *et al.*, 2017; Thines, 2011; Thines *et al.*, 2020; Voglmayr *et al.*, 2014). These organisms were most likely introduced with seeds or latently infected plants, making clear the necessity for better quarantine procedures for alien plants and for local production of plants and seeds whenever possible.

Status

Europe and Central Asia has a well-documented history of biological invasions by alien plant and animal parasitic fungi and oomycetes. Well-known examples are *Batrachochytrium dendrobatidis* (chytrid fungus; Longcore *et al.*, 1999), *Aphanomyces astaci* (crayfish plague; Mrugała *et al.*, 2015), *Phytophthora infestans* (Phytophthora blight; Yoshida *et al.*, 2013), and *Plasmopara viticola* (grapevine downy mildew; Gessler *et al.*, 2011). In addition, alien species have also invaded Europe as saprotrophsor symbionts, but the few documented examples such as *Clathrus archeri* (devil's fingers) are likely only the tip of the iceberg (Desprez-Loustau *et al.*, 2007; Litchman, 2010).

In Europe and Central Asia, the highest numbers of invasive alien forest pathogenic fungi are reported from the central-southern region (e.g., France, Italy, and Switzerland; Santini *et al.*, 2013). For example, *Phytophthora ramorum* (sudden oak death), which has had significant impacts on native forests, is thought to have been introduced to the United Kingdom *via* the ornamental plant trade (Jung *et al.*, 2021). Most forest pathogenic fungi are native to the northern hemisphere, but about one third are of unknown origin (Desprez-Loustau, 2009). The incidence in Europe of alien powdery mildews (Erysiphales) is higher in terms of expected species numbers and this may reflect responses to climate change in a group adapted for long-distance aerial spore dispersal (Heluta *et al.*, 2009). Using dried reference collection samples, Gross *et al.* (2021) demonstrated that three species of *Erysiphe* could be linked to the incidence of powdery mildew in oaks, a disease that emerged in Europe at the beginning of the twentieth century. By comparison, the incidence of specialized alien insect parasites of the order Laboulbeniales is comparatively low given their high species numbers (Desprez-Loustau, 2009). More aggressive genotypes of known plant pathogenic fungi may also arrive and become invasive (Arenz *et al.*, 2011). Alien and invasive microfungi pathogenic to animals include *Batrachochytrium dendrobatidis* (chytrid fungus), which is the agent of chytridiomycosis, a disease spread by trade and causing massive amphibian declines worldwide (Weldon *et al.*, 2004), and *Pseudogymnoascus destructans* (white-nose syndrome fungus) in bats (Thakur *et al.*, 2019).

Among all IPBES regions, Europe and Central Asia represents the region with the best available knowledge on the distribution of alien macrofungi with several national lists of alien fungi available (e.g., Desprez-Loustau *et al.*, 2010; Motiejūnaitė *et al.*, 2016). However, information for the Central Asian and Eastern European sub-regions, is much scarcer, and the absence or low number of alien macrofungi as known for these regions is likely a clear underestimation of actual numbers.

2.4.5.4 Islands

Mediterranean islands are biodiversity hotspots and have been invaded by large numbers of alien plant and animal species for centuries, many of which are now established (e.g., Capizzi, 2020; Chainho *et al.*, 2015; Ruffino *et al.*, 2009). Many North Sea and Baltic Sea islands have also been invaded, for example by *Mustela vison* (American mink) (e.g., Bonesi & Palazon, 2007). Islands belonging to Europe include overseas territories in most oceans. In particular, the United Kingdom and France have many islands in the southern Atlantic and in the Pacific. Biological invasions on islands related to European countries may be due to proximity of continents (islands off the Atlantic and Channel Sea coasts) or the colonization of more remote islands (e.g., French Polynesia and New Caledonia). Among the most studied taxa, the mammals of these islands, such as Gough Island, Crozet Island, or the Kerguelen Islands include rats, mice, cats, cattle, and mouflons (Davies *et al.*, 2015; C. W. Jones *et al.*, 2019; Pascal, 1980).

2.4.5.5 Data and knowledge gaps

While sampling and reporting intensity is high for alien mammals in Western Europe, data coverage and quality decrease eastward towards Eastern Europe, including Russia (Biancolini *et al.*, 2021). Significantly fewer sources of information are available for these areas in comparison to Western Europe and reports frequently lack extensive details on alien species trends, ecology, distribution, and impacts. This could reflect linguistic barriers that hinder data sharing (Angulo *et al.*, 2021) as the available literature published in English with respect to Eastern Europe cites numerous works written in other languages (e.g., Russian) (Khlyap *et al.*, 2011). A similar situation is reported for freshwater species, which are well reported for Europe, especially Western Europe (Nunes *et al.*, 2015), while less data are available for Central Asia.

While information available on alien amphibians and reptiles in this IPBES region has been thoroughly collected (Capinha *et al.*, 2017; Kark *et al.*, 2009; Kraus, 2009), some countries in Western Europe and Central Asia have been understudied and those lists of alien amphibians and reptiles are likely incomplete (Capinha *et al.*, 2017; Seebens, Blackburn, *et al.*, 2017; N. J. van Wilgen *et al.*, 2018).

Europe is amongst the best-researched continents for plant invasions (Pyšek, Hulme, *et al.*, 2020) and many regions in Central and Western Europe possess high quality data compared to other parts of the world (Lambdon *et al.*, 2008; Pyšek, Blackburn, *et al.*, 2017; Pyšek, Pergl, Dawson, *et al.*, 2020). Many countries have specialized catalogues and inventories with information going beyond the distribution of alien species (e.g., Celesti-Grapow *et al.*, 2009; E. J. Clements & Foster, 1994; Essl & Rabitsch, 2002; Klotz *et al.*, 2003; Preston *et al.*, 2002, 2004; Pyšek *et al.*, 2002; S. C. P. Reynolds, 2002). For Eastern Europe, there are data gaps and incomplete species lists for several countries including a large part of Russia (van Kleunen *et al.*, 2015, 2019). Work is currently underway to close this data gap (e.g., Leostrin & Pergl, 2021; Vinogradova *et al.*, 2018), and more species are likely to be identified as established alien species in Europe. Some countries in Central Asia also lack inventories (appendix 1 in Pyšek, Blackburn, *et al.*, 2017).

2.5 TRENDS AND STATUS OF ALIEN AND INVASIVE ALIEN SPECIES BY IPBES UNITS OF ANALYSIS

This section reports on the temporal trends and status of the distribution of alien and invasive alien species for each IPBES unit of analysis. IPBES units of analysis represent a broad-based global classification system considering both the state of nature in classes, equivalent to biomes, and in anthropogenically-altered biomes or "anthromes". The units correspond broadly to global classifications of nature and human interactions, serving the need for analysis and communication in a global policy context. More details about the units of analysis are provided in **Chapter 1**, **section 1.6.5** and online (IPBES, 2019b). The following section is sub-divided into an overview (**section 2.5.1**), terrestrial (**section 2.5.2**), freshwater (**section 2.5.3**), and marine (**section 2.5.4**) units of analysis as well as anthroponized areas (**section 2.5.5**).

2.5.1 Overview of trends and status by IPBES units of analysis

While no studies on biological invasion dynamics among comparative units of analysis exist, some studies have investigated patterns using similar delineations of study regions such as freshwater, marine, and terrestrial habitats. In general, far more studies are available for terrestrial alien species (although availability varies for above- and belowground) than for marine and freshwater systems. For instance, one comprehensive global analysis of first records of established alien species shows that 64 per cent of all studies had an explicit focus on terrestrial habitats, 13 per cent addressed marine and 12 per cent freshwater habitats, and the remaining were cross-taxonomic (Seebens, Blackburn, *et al.*, 2017). As a result, most established alien species have been reported from terrestrial habitats (over 75 per cent), while freshwater or marine alien species numbers are both of similarly low range (less than 10 per cent). Terrestrial alien species invasions were usually recorded earlier in time compared to freshwater species, which in turn were reported earlier than marine species (Zieritz *et al.*, 2017). Likewise, before 1840 most (about 75 per cent) established alien species recorded in north-western Europe represented terrestrial species, and the proportion has dropped continuously to less than 20 per cent more recently (Zieritz *et al.*, 2017). Only a few studies compared the trends and status of alien species across terrestrial, freshwater, and marine habitats at large spatial scales (e.g., Roy, Peyton, *et al.*, 2014; Sandvik, Dolmen, *et al.*, 2019; H. Xu *et al.*, 2012; Zieritz *et al.*, 2017). Other studies reported similar increases in established alien species across terrestrial, marine, and freshwater habitats with a tendency of freshwater alien species numbers accelerating more rapidly in recent years (O'Flynn *et al.*, 2014; Roy, Preston, *et al.*, 2014; H. Xu *et al.*, 2012).

2.5.2 Trends and status of alien and invasive alien species in terrestrial units of analysis

Box 2 7 **Mountain regions: A global assessment of trends and status of alien and invasive alien species.**

Elevational patterns of plant invasions have been described for many mountain regions around the world and with very few exceptions, established alien species richness peaks at lower elevations and declines towards the highest elevations, closely following patterns of human settlements and disturbance (e.g., Alexander *et al.*, 2011; Fuentes-Lillo *et al.*, 2021; Haider *et al.*, 2010; Pauchard *et al.*, 2009; Pérez-Postigo *et al.*, 2021; Tanaka & Sato, 2016). Most introduced alien species are pre-adapted to the environmental conditions at low elevations and need a broad environmental tolerance to spread towards high mountain sites (Alexander *et al.*, 2011). Therefore, alien species at high elevations are typically environmental generalists, and only rarely are mountain specialist species directly introduced at high elevations (Alexander *et al.*, 2016; Steyn *et al.*, 2017). As the regional lowlands are the most important source of alien plants found at high elevations, alien mountain floras are surprisingly dissimilar across mountain ranges and continents. In a study analyzing alien species lists from 13 mountain regions, about 60 per cent of alien species were recorded in a single mountain area, and less than 5 per cent were found in more than half of the regions included in the study (McDougall *et al.*, 2011).

Anthropogenic corridors such as roads, trails, and railways strongly facilitate the spread of alien plants from low to high elevations (Alexander *et al.*, 2011; Lembrechts *et al.*, 2017; Liedtke *et al.*, 2020; Rashid *et al.*, 2021; M. Yang *et al.*, 2018), and alien plants are much more common in disturbed habitats directly adjacent to such corridors compared to more remote natural habitats (Seipel *et al.*, 2012). Thus far, few alien species have been able to penetrate natural communities, especially at higher elevations, but those that have invaded are often shade and moisture tolerant (McDougall *et al.*, 2018).

While there is no evidence that alien species in mountains have caused the local extinction of native species, they have a strong impact on multiple dimensions of biodiversity (B. W. van Wilgen *et al.*, 2020). First, they reduce differences in community composition between low and high elevations, and thus negatively affect beta-diversity, leading to a biotic homogenization in mountains – and in the long-term maybe also across mountain regions. A global study based on a standardized vegetation survey demonstrated that alien species along roadsides either shifted the richness peak of native plants to lower elevations, or even changed the shape of the relationship between native species richness and elevation (Haider *et al.*, 2018).

In the last 15–20 years, research on plant invasion patterns in mountains has increased markedly. However, published studies are unevenly spread across mountains worldwide. While there are many studies from regions with temperate or Mediterranean climates, there are few from the subtropics and tropics (e.g., the Andes, mesoamerica, Africa, and Asia) or high latitude boreal and Arctic regions. A second shortcoming is the lack of long-term monitoring of alien species in mountains. Few studies have used permanent monitoring sites to document changes in alien species occurrence in mountains (but see Kalwij *et al.*, 2015; Turner *et al.*, 2021). The Mountain Invasion Research Network (MIREN, www.mountaininvasions.org) has developed a standardized survey protocol to study and monitor patterns of plant invasions into mountains (but not in Africa), which has been applied in 19 regions worldwide since 2007 (Haider *et al.*, 2022; **Figure 2.34**). While assessing future trends of alien plant species distributions in mountains remains a challenge, efforts are being conducted to model invasions using data collected at multiple scales especially under climate change (Lembrechts *et al.*, 2017; Petitpierre *et al.*, 2016) and shifts in biotic interactions using evidence collected through both observational and experimental approaches. Such studies show that future plant invasions in mountains will increase in the future under climate change and increased anthropogenic pressure (Alexander *et al.*, 2016; Petitpierre *et al.*, 2016).

2.5.2.1 Tropical and subtropical dry and humid forests

Tropical and subtropical forests cover about 52 per cent of global forested land and hold 200 billion tons of carbon in aboveground biomass (IPBES, 2019a). These ecosystems harbour the highest biological diversity globally, but also the highest number of threatened species (IPBES, 2019a). Since 1990, over 250 million hectares were cleared for agriculture and urban expansion, infrastructure and mining (IPBES, 2019a; Vancutsem *et al.*, 2021). Although some regions have reported net gain in forest cover, this trend is mainly driven by planted-forest expansion with alien tree and palm species (Sloan & Sayer, 2015).

Trends

Historically, tropical and subtropical dry and humid forests have experienced fewer introductions of alien species relative to temperate ecosystems. Compared to other mainland terrestrial regions of the globe, tropical and subtropical dry and humid forests have lower numbers of

Box 2 7

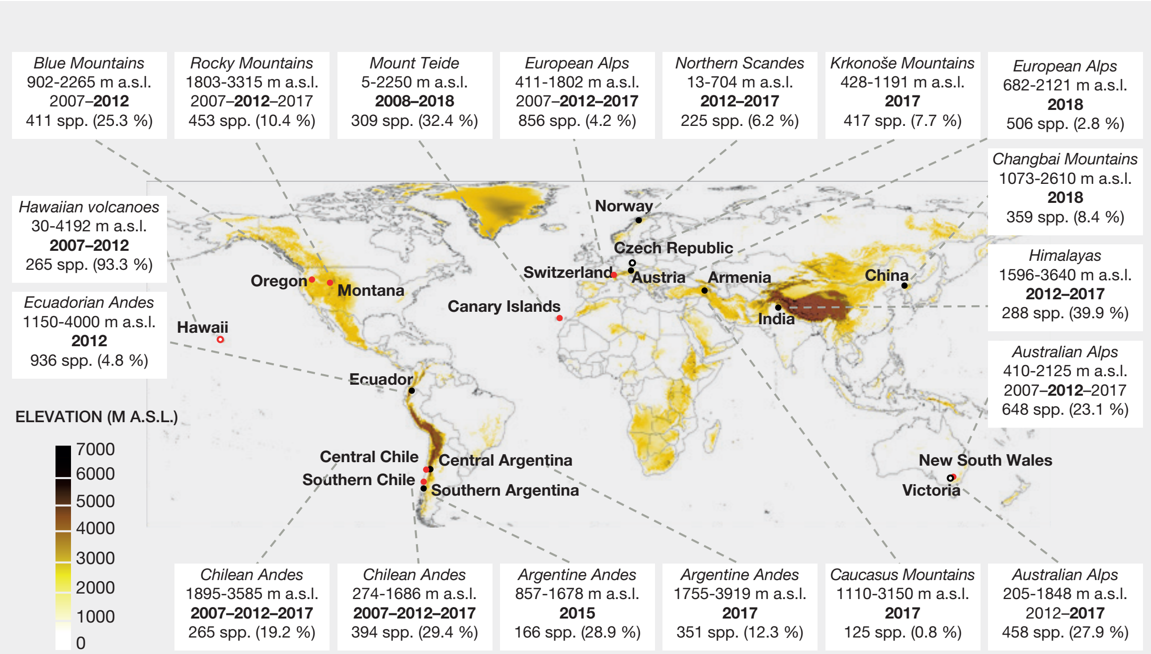

Figure 2 34 **Locations of the MIREN surveys.**

Sites in mountain regions where MIREN surveys have been used to sample and monitor plant invasions across elevational gradients. Source: Haider *et al.* (2022), https://doi.org/10.1002/ece3.8590, under license CC BY 4.0.[3]

invasive alien species for all taxonomic groups (Dawson *et al.*, 2017). For instance, records of invasive alien species in the tropical and dry forests of South America mostly date from the past 50 years and have increased only in the last 20 years (Zenni, 2015; Zenni & Ziller, 2011). Also, tropical South America has two or three times fewer established alien plants than temperate South America despite its greater area (Zenni *et al.*, 2022). However, the recent and ongoing increases in biological invasions in tropical and subtropical dry and humid forests can be attributed in large part to agricultural and urban expansion and increased propagule pressure (Waddell *et al.*, 2020). Forest degradation and clearcutting allow the establishment and spread of numerous invasive alien grass species, some of the most prominent invaders in tropical forest ecosystems (Dar *et al.*, 2019; Zenni, 2015; Zenni & Ziller, 2011).

Lack of reliable baseline information from most countries in Asia prevents a comprehensive analysis of trends of alien plant invasions in tropical and subtropical forests in this region. Available information shows an increase of one to eight major species during a period of 7-18 years in five countries in the region (Banerjee *et al.*, 2021; Government of Myanmar, 2005; Islam *et al.*, 2003; Khuroo *et al.*, 2012; Mukul *et al.*, 2020; Pallewatta *et al.*, 2003; Shrestha & Shrestha, 2021; Tiwari *et al.*, 2005; Wijesundara, 2010).

Status

Some tropical and subtropical dry and humid forests on islands have some of the most noteworthy examples of biological invasions. Hawaii, for instance, has a greater number of established alien species than native species (G. W. Cox, 1999). Species such as *Psidium cattleianum* (strawberry guava), *Morella faya* (firetree), *Hedychium* spp. (ginger), and *Sus scrofa* (feral pig) have caused significant ecological impacts in Hawaiian tropical forests. Another highly invaded tropical island, the Galapagos, considers biological invasions the most relevant threat to native

3. This map is directly copied from its original source (Haider *et al.*, 2022) and was not modified by the assessment authors. The map is copyrighted under license Attribution 4.0 International (CC BY 4.0). The designations employed and the presentation of material on the maps used in the assessment do not imply the expression of any opinion whatsoever on the part of IPBES concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared or used for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein and for purposes of representing scientific data spatially.

biodiversity and the alien taxa outnumber the native species (Zenni *et al.*, 2022). In Guam, invasive alien reptiles (notably *Boiga irregularis* (brown tree snake)) and some invasive alien tree species have been reported to extirpate native species and drastically change ecosystem processes (Fritts & Leasman-Tanner, 2001; Marler, 2020).

In South America, there are 247 known established alien plant species in Bolivia, 503 in Brazil, 265 in Colombia, 348 in Ecuador, 166 in Guyana, 72 in Paraguay, 288 in Peru, and 219 in Venezuela (Zenni *et al.*, 2022). For the Caribbean, there are at least 446 invasive alien species known among plants, invertebrates, vertebrates, fungi, and diseases (Kairo *et al.*, 2003). *Herpestes javanicus auropunctatus* (small Indian mongoose) is one of the most notorious of these species in the Caribbean as it has been associated with the extinction of five native species. In Asia, 179 invasive alien species have been recorded in tropical forests of central India (Dar *et al.*, 2019). For plants, the numbers of invasive alien plants in tropical and subtropical forests (based on data from 10 countries) range from 15 to 58, the highest being in forests of Indonesia (58 species) followed by forests in China (52) (Banerjee *et al.*, 2021; Mukaromah & Imron, 2019; Mukul *et al.*, 2020; Qureshi *et al.*, 2014; Shrestha & Shrestha, 2021; D. T. Tan *et al.*, 2012; Weber *et al.*, 2008; Wijesundara, 2010; H. Xu *et al.*, 2012). The most widespread species in the region are *Lantana camara* (lantana) (recorded in 18 countries of the 19 for which data are available), *Leucaena leucocephala* (leucaena, 18 countries), *Mikania micrantha* (bitter vine, 16 countries), *Ageratum conyzoides* (billy goat weed, 16 countries), *Chromolaena odorata* (Siam weed, 15 countries), *Mimosa diplotricha* (giant sensitive plant, 13 countries), *Prosopis juliflora* (mesquite, 12 countries) and *Parthenium hysterophorus* (parthenium weed, 11 countries).[4] In India, the invasive alien plant *Chromolaena odorata* dominates the understory of forests and has been shown to negatively affect the pollination of native species (Peh, 2010; **Chapter 4**, **section 4.4.3**). Another invasive alien plant *Lantana camara*, a plant species native to South America and invasive in most tropical regions of the world, can greatly reduce the productivity of economically important plants (Peh, 2010).

In Africa in recent decades the establishment of alien tree plantations, mainly pines and eucalyptus, has been a high priority in governmental forestry (Obua *et al.*, 2010; Tumushabe & Mugyenyi, 2017). The replacement of natural forests with alien species, coupled with other human disturbances, has compounded the threat of invasive alien species that include plants such as *Broussonetia papyrifera* (paper mulberry), *Senna spectabilis* (whitebark senna), *Lantana camara* (lantana; Totland *et al.*, 2005), and also insect species like *Gonometa podocarpi* (podocarpus moth; FAO, 2012), *Achaea catocaloides* (African apple tree moth; e.g., Martins *et al.*, 2015) and *Leptocybe invasa* (blue gum chalcid; FAO, 2012). These invasive alien species have the potential to pose a threat to forest ecosystems (Hamilton *et al.*, 2016). However, very little is known about the invasion of alien species into tropical forests and there is no up-to-date detailed assessment of the potential risks that these invasive alien species, especially under rapidly changing climate, are causing to the forests and their associated biodiversity and nature's contributions to people (**Chapter 3**, **section 3.3.4**).

4. Data extracted from the Global Invasive Species Database (GISD; http://www.iucngisd.org/gisd/), GRIIS (https://doi.org/10.5281/zenodo.6348164) and Association of South-east Asian Nations (ASEAN; https://asean.org/)

Data and knowledge gaps

A worldwide review of invasive alien species in tropical and subtropical dry and humid forests has never been done, and most data available to date are at the country-level rather than at the level of biogeographic regions such as units of analysis. Of the countries with major areas covered by tropical and subtropical dry and humid forests, data are available mostly for South America, some parts of Mesoamerica and the Caribbean, and for South Asia, while data is scarce for tropical and subtropical dry and humid forests in Africa.

Biological invasions in tropical and subtropical dry and humid forests have been less studied than most other terrestrial ecosystems. This lack of data is, in part, explained by the lower numbers of invasive alien species recorded for tropical forests compared to other ecosystems. However, given the growing anthropogenic pressure over these regions, it is likely that biological invasions will increase in the next decades in tropical and subtropical forests, especially in regions with high intensity of land use change. Most reports available for tropical and subtropical dry and humid forests are for plant invasions, and there is very limited data on animal invasions except for a few well-studied species, such as *Herpestes javanicus auropunctatus* (small Indian mongoose) and *Boiga irregularis* (brown tree snake). For most regions with these forests, lists of established plant species are available (Pyšek, Pergl, *et al.*, 2017; van Kleunen *et al.*, 2019), but these data provide very little insight into the actual situation of biological invasions in tropical and subtropical dry and humid forests (e.g., spread and impacts).

As a general trend in Asia, the cumulative number of invasive plants is known to increase exponentially over years (e.g., in China: H. Xu *et al.*, 2012). However, information on trends and status of invasive alien plants in tropical and subtropical forests in Asia are largely unavailable. Attempts are currently being made by some countries to prepare national inventories for invasive alien plants (e.g., Dorjee *et al.*, 2020; Mukul *et al.*, 2020), though these lists do not appear to include information on the habitats in which the alien species occur.

2.5.2.2 Temperate and boreal forests and woodlands

Trends

The view that forested ecosystems are resistant to invasions by alien plants has eroded over the past two decades as observations of local dominance by both herbaceous and woody invaders in forests worldwide accumulate (Fridley, 2013; Liebhold *et al.*, 2017; P. H. Martin *et al.*, 2009). Although estimates of trends in alien plant richness specific to forests are difficult to determine for most regions, biological invasions in temperate forests are increasing globally and will likely accelerate as high latitudes continue to warm with climate change (Pauchard *et al.*, 2016; **Chapter 3**, **section 3.3.4**), particularly for boreal forests (Mulder & Spellman, 2019; Sanderson *et al.*, 2012). Habitat fragmentation and road-building activities are also principal drivers that facilitated he increase in forest plant invasions (**Chapter 3**, **section 3.3.1.2**), both as a means to disperse alien propagules and to increase light and nutrient availability, which facilitate the growth of invader source populations that may spread into adjacent closed-canopy forests (R. O. Bustamante & Simonetti, 2005; Flory & Clay, 2009; Kuhman *et al.*, 2010). Afforestation (i.e., plantation of trees in areas without previous tree cover) represents another driver that promotes biological invasions (Ramprasad *et al.*, 2020). Forest invasion research lags behind that of grasslands and wetlands (Nunez-Mir *et al.*, 2017), and temperate and especially boreal forests tend to be remote, making the early stages of biological invasions difficult to monitor (Liebhold *et al.*, 2017). As a result, the colonization of temperate and boreal forests by alien plants is likely much greater than currently reflected in the literature (P. H. Martin *et al.*, 2009).

Status

In the Northern Hemisphere, North American deciduous forests have a larger number of alien plant species than those of Europe and Asia (Fridley, 2013; Heberling *et al.*, 2017), including a substantial number of alien shrubs, lianas, and small trees introduced as ornamentals (Fridley, 2008). In contrast, the most negatively impactful alien plants in European temperate forests are trees (**Chapter 4**, **section 4.3.2.1**; Campagnaro *et al.*, 2018; Essl *et al.*, 2011; Langmaier & Lapin, 2020), many of which were intentionally introduced for timber production or forest reclamation (e.g., *Prunus serotina* (black cherry; Closset-Kopp *et al.*, 2007), *Quercus rubra* (northern red oak; Major *et al.*, 2013), *Robinia pseudoacacia* (black locust; Vítková *et al.*, 2017)), and woody species are the most numerous species in forest understory (V. Wagner *et al.*, 2017). Deciduous forests of East Asia, which tend to have higher levels of native species richness than other temperate forests (Qian & Ricklefs, 2000), remain relatively uninvaded (B. Auld *et al.*, 2003; Fridley, 2013; but see Wavrek *et al.*, 2017); further, woody species in general are strongly under-represented in the alien floras of China (Axmacher & Sang, 2013; Weber *et al.*, 2008), Korea (Heberling *et al.*, 2017), Japan (B. Auld *et al.*, 2003), and the Russian Far East (Kozhevnikov & Kozhevnikova, 2011). Boreal forests across the northern hemisphere are among the least invaded forest types outside the tropics (Leostrin & Pergl, 2021; Sanderson *et al.*, 2012); however, climate change is widely expected to accelerate understory plant invasions (Mulder & Spellman, 2019; **Chapter 3**, **section 3.3.4**), and many fast-growing herbaceous alien species are already disrupting native tree regeneration in forest gaps (e.g., *Cirsium arvense* (creeping thistle); Humber & Hermanutz, 2011). In European (deciduous) forests, 386 alien plant species were recorded in forest understory and the most common, *Impatiens parviflora* (small balsam), was recorded in 21 per cent of sampled plots (V. Wagner *et al.*, 2017). Plant invasions of forests of temperate South America remain understudied but there is some evidence that North American plantation conifers (e.g., *Pinus contorta* (lodgepole pine), *Pseudotsuga menziesii* (Douglas-fir)) are able to establish in native evergreen forests (Pauchard & Alaback, 2004; Peña *et al.*, 2008; Simberloff *et al.*, 2009), along with herbaceous species such as *Prunella vulgaris* (self-heal; Godoy *et al.*, 2011). Plantation conifers (e.g., *Pinus radiata* (radiata pine)) are also an increasing concern in dry eucalypt forests of Australia (M. C. Williams & Wardle, 2005).

Data and knowledge gaps

Although alien plant lists are increasingly available for regions where forest invasions are understudied, including Turkey (Akbulut & Karaköse, 2018; Yazlık *et al.*, 2018), Iran (Sohrabi *et al.*, 2021), and Siberia (Vinogradova *et al.*, 2018), the richness and abundance of invasive alien plants specific to temperate forested habitats remains unknown for many regions outside North America and Europe (Heberling *et al.*, 2017). One of the key knowledge gaps is the role of shade tolerance in alien species establishment: many alien plants establish following disturbance and persist under a closed canopy, but relatively few alien plants can recruit into intact temperate and boreal forests (P. H. Martin *et al.*, 2009; V. Wagner *et al.*, 2021). A priority of future research is to understand the interplay of disturbance, climate change, and biological invasions (**Chapter 3**, **section 3.3.4**) in altering the trajectory of native forest stands to what will likely become novel communities of mixed native and alien species (Chmura, 2020).

2.5.2.3 Mediterranean forests, woodlands and scrub

Trends

Although no comprehensive analysis of the trends of alien species for Mediterranean ecoregions (Mediterranean Basin, South Africa, North America, South America and Australia) exists, it seems likely that the number of alien species increases as observed for other regions. As with other units of analysis, increases in the number of alien species and rates of new records results not only from increased transport of species (e.g., trade, human population, spread, tourism; M. C. Jackson & Grey, 2013), but also from increasing wildfires (e.g., Keeley *et al.*, 2005), increased sampling intensity (both in the field and for bibliographic searches) and greater awareness of invasive alien species (L. Henderson & Wilson, 2017). Some regions and taxa have recently shown a deceleration in new introductions as a result of successful invasive alien species management or national and transnational regulations (European Union, 2014; Murray & Phillips, 2012). This is the case with, for example, birds in the Iberian Peninsula (Abellan *et al.*, 2016), plants and terrestrial vertebrates in Chile (Fuentes *et al.*, 2020), and invasive plants in Australia (Murray & Phillips, 2012).

In South Africa, the South African Plant Invaders Atlas reports a general increase in both the numbers of alien plant species and total area occupied (L. Henderson, 2007). While the rate of spread of alien plants decreased in some cases and even contracted in a few cases as a result of classical biocontrol, overall, 172 new alien plant species emerged between 2006 and 2016 and those already established expanded their ranges (L. Henderson & Wilson, 2017). An increase in alien species numbers in the Mediterranean parts of the country, due to horticulture and floriculture, is reported; the area of fynbos in South Africa is referred to as one the most heavily invaded biomes in the country (L. Henderson, 1998; B. W. van Wilgen, 2018).

Some countries in the Mediterranean Basin (e.g., Portugal) have good records of temporal trends of plant species dating back to 1500. A steady increase in alien species numbers occurred over time with an acceleration in the introduction of new species at late nineteenth century, some deceleration between 1930–1940 and a new acceleration at least up to 2018 (Almeida, 2018; Almeida & Freitas, 2001). Other countries in the Mediterranean Basin, such as Albania (Barina *et al.*, 2014), experienced accelerated introductions later during the mid-twentieth century with few alien species reported before that time.

From 1500 to 1903 more populations of alien birds were introduced to the Mediterranean parts of South Africa, Australia, California, and fewer to Chile and the north-western countries of the Mediterranean Basin. By the end of the twentieth century, this trend exhibited some changes with more bird populations introduced in the north-western countries of the Mediterranean Basin (with a hotspot in Spain), in Western Cape (South Africa) and California (United States) (E. E. Dyer, Cassey, *et al.*, 2017). At least in the Iberian Peninsula, the pronounced increase after 1955 – particularly steep after the 1980s – was followed by a decrease by 2005, possibly explained by the ban of wild-caught birds in Spain after the avian flu and regulations to reduce invasion risk (Abellan *et al.*, 2016).

Amphibians and reptiles were reported as introduced to Mediterranean areas only after 1800, with increasing numbers of records of new established alien species after mid-1900 (Capinha *et al.*, 2017).

In California, United States, alien terrestrial macroinvertebrates have been established since 1700, with many species (ca. 39 per cent) introduced before 1930. A more detailed analysis from 1935 – 2010 demonstrates the regular detection of new species of alien arthropods across the 75 years in three distinct phases: higher mean values early in this period, decreased detections 1970 to late 1980s, followed by an increase (Dowell *et al.*, 2016).

Status

Comprehensive information about terrestrial alien vascular plants is available for most countries with a Mediterranean climate (e.g., Almeida, 2018; Arianoutsou *et al.*, 2010; Barina *et al.*, 2014; Fuentes *et al.*, 2020; Galasso *et al.*, 2018; Meddour *et al.*, 2020; B. W. van Wilgen, 2018), and most of the checklists provide information about the status of the species (Pyšek, Pergl, *et al.*, 2017 for summary data on established alien plants).

All the Mediterranean regions share a higher percentage of alien plant species with southwest Australia than with any other region. Chile and the Mediterranean Basin share comparatively fewer alien plant species with the other regions (Arianoutsou *et al.*, 2013). Common invasive plants in and from Mediterranean areas are *Oxalis pes-caprae* (Bermuda buttercup), *Acacia* spp., *Carpobrotus edulis* (hottentot fig), *Ulex* spp. (Gorse), *Cytisus* spp., and *Hakea* spp. (Pincushion tree). Most Mediterranean areas also share alien species that have originated from different climates, e.g., *Ailanthus altissima* (tree-of-heaven), *Conyza* spp., and *Agave americana* (century plant).

Publications on alien plants are more common than for other taxonomic groups (e.g., Chile; Fuentes *et al.*, 2020; N. J. van Wilgen *et al.*, 2018; IUCN SSC Invasive Species Specialist Group (ISSG)). In Mediterranean areas, alien bird species richness is high in some regions of California, western parts of the Mediterranean basin, South Africa, and Australia (E. E. Dyer, Cassey, *et al.*, 2017). Alien reptiles and amphibians (Capinha *et al.*, 2017) present in the five global Mediterranean areas are more numerous in terms of species numbers in California and Spain, and have few documented species (or are even absent) in northern Africa and Eastern Europe. Terrestrial invertebrates also show high numbers of alien species, for example, in California (over

1,600 species, approximately 85 per cent insects) (Dowell *et al.*, 2016).

Data and knowledge gaps

In countries covering multiple units of analysis, the trends and status for alien species in the Mediterranean zone is mostly not specifically described. Some countries with Mediterranean climates, particularly Syria, Lebanon, Malta, and Macedonia, have not yet published comprehensive inventories of alien species. Detailed distribution maps of specific alien species in Mediterranean areas are not frequently found.

2.5.2.4 Arctic and mountain tundra

Trends

Early introductions of alien plant and vertebrate species in polar regions were largely intentional (e.g., revegetation of industrial sites and fur farming (Forbes & Jefferies, 1999; Usher, 2005), while current introductions are often unintentional (Tolvanen & Kangas, 2016; Wasowicz *et al.*, 2020). Future increases in alien species richness across taxonomic groups for both Arctic and mountain tundra regions is expected due to climate change and increasing anthropogenic activity including deliberate ornamental plant introduction related to tourism development or unintentional introductions along roads, trails, and mineral extraction sites (**Chapter 3**, Carboni *et al.*, 2018; Nielsen & Wall, 2013; Normand *et al.*, 2013; Petitpierre *et al.*, 2016; Solovjova, 2019; C.-J. Wang *et al.*, 2017; Ware *et al.*, 2016; Wasowicz *et al.*, 2013). However, a modelling study on the 100 world's worst invaders projected no increase in suitability of tundra regions to invasive alien species until 2100 as climatic conditions for some of these species might become too extreme in the future, or as ongoing degradation and land use change might render current habitats unsuitable (Bellard, Thuiller, *et al.*, 2013). Invasive alien disease risks are likely to increase in the future under climate change, with potential increases in disease transmission between domestic species and Arctic wildlife, as well as through increased survival probability and range expansion of introduced disease vectors or increased host susceptibility under climate change (Bradley *et al.*, 2005; Dudley *et al.*, 2015; Kutz *et al.*, 2004; Waits *et al.*, 2018).

Similarly, mountain regions have been mostly spared from biological invasions because of low anthropogenic pressure and harsh climates (Kueffer *et al.*, 2013; Pauchard *et al.*, 2009; Petitpierre *et al.*, 2016). However, many high mountain regions globally have increasing alien species richness, especially for plants (Alexander *et al.*, 2016; Becker *et al.*, 2005; Carboni *et al.*, 2018; Pauchard *et al.*, 2009; Pickering *et al.*, 2007; Williamson & Fitter, 1996). Future alien species colonizers are expected to have wide climatic niches (like most current invasive alien species) and will likely increase their range sizes from low elevations *via* an upward expansion of their current range limits, with expansion rates for alien plants being twice as high as for native plant species (Alexander *et al.*, 2011, 2016; Carboni *et al.*, 2018; Dainese *et al.*, 2017). Direct introductions of more specialized (i.e., cold adapted) alien species into high elevation environments will also likely increase because of increased tourism and targeted introduction for ornamental purposes (Alexander *et al.*, 2016; Carboni *et al.*, 2018; Godde *et al.*, 2000; Kueffer *et al.*, 2013; McDougall *et al.*, 2005). Genetic adaptability of alien species at range margins resulting in the colonization of cooler sites will likely further increase the risk of future invasions (Alexander, 2010). Bryophytes are common alien species in cold environments (Rozzi *et al.*, 2008) and the likelihood of alien bryophytes invading high mountain and Arctic tundra ecosystems is assumed to be high (Essl *et al.*, 2013; Pauchard *et al.*, 2016).

Status

Established alien species richness across taxonomic groups decreases towards higher latitudes (Capinha *et al.*, 2017; E. E. Dyer, Cassey, *et al.*, 2017; Essl *et al.*, 2013; Pyšek & Richardson, 2006; Qian, 2008; Sax, 2001) and high elevations (M. Ahmad *et al.*, 2018; Alexander *et al.*, 2011; Q. Guo *et al.*, 2021; Haider *et al.*, 2010; Kalwij *et al.*, 2008; Khuroo *et al.*, 2011; Marini *et al.*, 2013; Western & Juvik, 1983), but exceptions exist (Paiaro *et al.*, 2011; Rosa, 2020). Arctic regions have been identified as coldspots for alien species richness across different taxonomic groups (e.g., plants, birds, mammals, spiders, ants, amphibians, reptiles, fishes), especially Greenland, northern North America and northern Europe (Dawson *et al.*, 2017). Alaska and northern Central Asia have higher alien richness of several taxonomic groups, but these patterns might be influenced by different sampling intensity and data availability across regions (Dawson *et al.*, 2017). In mountain and arctic tundra, alien plants are generally found in anthropogenically disturbed sites and along transportation infrastructure routes (Alexander *et al.*, 2011, 2016; Forbes & Jefferies, 1999; Haider *et al.*, 2010; Kalwij *et al.*, 2008; Khuroo *et al.*, 2011), and their richness decreases with increasing distance from these structures (Arteaga *et al.*, 2009; Haider *et al.*, 2022; Pauchard & Alaback, 2004; Seipel *et al.*, 2012). Successful invaders are mainly graminoid or weedy species (Alexander *et al.*, 2016; Carey *et al.*, 2016; Forbes & Jefferies, 1999; Wasowicz *et al.*, 2020) however, primary invasion along mountain roads tends to promote longer lived species (McDougall *et al.*, 2018). Species richness increases across taxonomic groups are mainly linked to invasions from lower elevations and latitudes under climate change, and increasing anthropogenic pressure associated with intentional introductions (Alexander, 2010; Bertelsmeier *et al.*, 2015; Carboni *et al.*, 2018; Dainese *et al.*, 2017;

Godde *et al.*, 2000; Greve *et al.*, 2017; Kueffer *et al.*, 2013; McDougall *et al.*, 2005; Parkinson & Butler, 2005; Wasowicz *et al.*, 2013, 2020) but some invasive alien species might also lose suitable habitats when the climatic conditions become too extreme in the future (Bellard, Thuiller, *et al.*, 2013).

Data and knowledge gaps

No dedicated gap analysis is currently available for Arctic and mountain tundra regions. However, the same regional gaps emerge across taxonomic groups as for global alien richness datasets. In particular, data is missing for most taxonomic groups in the northern part of Asia (Dawson *et al.*, 2017) and research efforts are generally less intensive for animals and plants at higher latitudes (Lenoir & Svenning, 2015). Given that animals and plants are two of the most studied taxonomic groups, this is likely also true for other taxonomic groups such as mosses, lichens, and microorganisms.

2.5.2.5 Tropical and subtropical grasslands

In the Millennium Ecosystem Assessment (2005) tropical grasslands and savannas were regarded as less affected by plant invasions relative to other biomes, but there is an increasing trend in both distribution and alien species richness in these biomes. Thus, although invasive alien species have only recently been considered as a main threat to biodiversity conservation and functioning of tropical grasslands and savannas, they are likely to become much more widespread in the future. Within the grassland-savanna biome, frequently seasonally flooded river and stream banks are generally substantially more vulnerable to plant invasions than areas away from rivers (Pyšek, Hulme, *et al.*, 2020; D. M. Richardson *et al.*, 2007), but with notable exceptions.

The current low incidence and impact of alien plants in savannas relative to some other terrestrial biomes may be because disturbance, which generally favours invasions, is fundamental to savanna functioning (**Chapter 4**, **section 4.3.2.1**). Savannas are resilient to changes in disturbance regimes (Harrison & Shackleton, 1999; Walker & Noy-Meir, 1982), making them relatively resistant to biological invasions in some areas (Foxcroft, Richardson, *et al.*, 2010). Drivers facilitating plant invasions in savannas include herbivore presence, residence time, intentional introductions for pasture improvements, the introduced species' physiology, and anthropogenic disturbance (Foxcroft, Richardson, *et al.*, 2010). While fire regimes may play a role in preventing alien plant invasions in fire prone systems, the increasing invasion of cacti (less affected by fire in areas denuded of grass cover) in African savannas, and fire adapted African grasses in northern Australian and southern American savanna grasslands are overcoming this barrier.

Trends

Although no study about trends of alien species in tropical and sub-tropical grasslands yet exists, it seems likely that the number of alien species are increasing likewise to other regions worldwide such as temperate grasslands (**section 2.5.2.6**).

Status

Foxcroft, Richardson, *et al.* (2010) suggested that African savannas are less invaded than savannas in the Neotropics and northern Australia, where alien African grasses especially have had significant impacts, due to (i) lower rates of intentional plant introductions to that continent, (ii) the role of large mammalian herbivores in African savannas, (iii) historical and biogeographical issues relating to the regions of origin of alien species, and (iv) the adaptation of African systems to fire. Moreover, many forms of anthropogenic land use over a long period (Bourlière & Hadley, 1983), together with high levels of frequent disturbances, may have resulted in alien plants being not yet very widespread or common in African savannas (Foxcroft, Richardson, *et al.*, 2010). In Southern Africa, L. Henderson and Wells (1986) listed 583 established alien plants for tropical savannas, of which 151 were known to be particularly impactful invasive alien species, and L. Henderson (2007) reported 48 alien species for the savanna biome of South Africa alone. *Lantana camara* (lantana), *Chromolaena odorata* (Siam weed) and *Melia azedarach* (Chinaberry) were the most prominent invasive alien species, followed by *Solanum mauritianum* (tobacco tree), *Acacia mearnsii* (black wattle), *Opuntia ficus-indica* (prickly pear), *Ricinus communis* (castor bean), *Psidium guajava* (guava), and *Jacaranda mimosifolia* (jacaranda). Examples of invasive alien species in protected areas include *Chromolaena odorata* in Hluluwe-Imfolozi (Macdonald, 1983) and *Opuntia stricta* (erect prickly pear) in Kruger National Park (Foxcroft *et al.*, 2004). More recent evidence from East Africa suggests these trends of savannas being less invaded are reversing and biological invasions are rapidly increasing. While the Serengeti-Mara ecosystem in East Africa is relatively free of widespread and abundant invasive alien plants, with a few exceptions, Witt *et al.* (2017) report 51 established alien plant species, with 21 of these recorded as invasive. They consider *Parthenium hysterophorus* (parthenium weed), *Opuntia stricta, Tithonia diversifolia* (Mexican sunflower), *Lantana camara, Chromolaena odorata*, and *Prosopis juliflora* (mesquite) to pose the greatest threats. In central Kenya, Laikipia County, which comprises grasslands, savanna woodland and forest, 145 alien plant species recorded, 67 and 37 were already established or invasive, respectively (Witt *et al.*, 2020). Widespread species in the county included *Opuntia stricta,* *Opuntia ficus-indica, Austrocylindropuntia subulata* (Eve's needle cactus), and other succulents (Witt *et al.*, 2020).

"New World" neotropical savannas are locally highly invaded mostly by African C4 grasses introduced for forage quality improvement (e.g., *Hyparrhenia rufa* (jragua grass), *Urochloa eminii* (signal grass), *Melinis minutiflora* (molasses grass), *Andropogon gayanus* (tambuki grass), *Panicum maximum* (Guinea grass); Rejmánek *et al.*, 2013). In Brazil, this practice was encouraged into the late 1990s (Pivello *et al.*, 1999). In Colombia, Venezuela, and Brazil, about 4 million km^2 were transformed to pasture by using, to a large extent, African C4 grasses (D. G. Williams & Baruch, 2000). Gorgone-Barbosa *et al.* (2015) also reported *Urochloa brizantha* (palisadegrass) to be an aggressive invasive alien grass in the Brazilian Cerrado. Trees are, however, also invasive in grassland savanna in São Paulo State, Brazil, where De Abreu and Durigan (2011) reported that *Pinus elliottii* (slash pine) has completely altered the structure of grassland savannas.

African and European grasses are common alien species in Australia (D'Antonio & Vitousek, 1992). Lonsdale (1994) reported that 466 alien pasture species were intentionally introduced into the savannas of northern Australian and many have become invasive (ca.13 per cent). The most impactful invasive alien species in Australian tropical savannas include *Andropogon gayanus* (Tambuki grass) introduced as a pasture grass in the 1930s, whose invasion has led to several-fold increases in the fuel load and fire intensity, further promoting this species' invasion (Rossiter *et al.*, 2003). In Kakadu, *Mimosa pigra* (giant sensitive plant), *Hymenachne amplexicaulis* (hymenachne), *Urochloa mutica* (para grass) (Setterfield *et al.*, 2013), *Cenchrus ciliaris* (buffel grass), *Cenchrus polystachios* (mission grass), *Themeda quadrivalvis* (grader grass) are other fire-regime altering African grasses, while *Vachellia nilotica* (gum arabic tree) from Africa, *Cryptostegia grandiflora* (rubber vine) from Madagascar, *Jatropha gossypiifolia* (bellyache bush) from Mesoamerica, *Lantana camara* (lantana) from the Neotropics, *Mimosa pigra* from South America, or *Prosopis* species (mesquite) from Americas, and *Ziziphus mauritiana* (jujube) from India are examples of woody species invading Australian savannas. There are also several cactus species introduced from Meso- and South America (Foxcroft, Richardson, *et al.*, 2010). Ratnam *et al.* (2019) also shows that across large stretches of fine- and broad-leaved savannas in Asia, *Lantana camara* and *Prosopis juliflora* are widespread, expanding widely over the past three to four decades.

Data and knowledge gaps

Tropical and subtropical savannas and grasslands are in regions understudied compared to other regions of the world making information about alien species scarce and comprehensive studies lacking. It therefore remains unclear to what degree the often-low numbers of reported established alien species in these ecosystems represent low research effort or true numbers. However, given the low numbers of available studies, it seems likely that numbers of established alien species are likely to be considerably higher than reported.

2.5.2.6 Temperate grasslands

Temperate grasslands once covered 5–10 per cent of the terrestrial surface (Dixon *et al.*, 2014; White *et al.*, 2000), yet now rank among the most threatened biomes globally due to land conversion and degradation (Hoekstra *et al.*, 2004; IPBES, 2019a). In North America, ca. 70 per cent of the Great Plains prairie have been converted to cropland and to a lesser degree to pastures and human settlements. Intensive grazing and agricultural usage have transformed many Pampas areas of South America. Conversion is also pronounced in some parts of Central Asia (including Kazakhstan, Kyrgyzstan, Russia, Tajikistan; V. Wagner *et al.*, 2020), but less so in highly continental Asia (Mongolia and China) where the world's largest temperate grasslands are still found (Wesche *et al.*, 2016).

Trends

The ongoing intensifying anthropogenic pressures on grassland ecosystems including climate change will likely further accelerate the establishment of new alien species in temperate grasslands (**Chapter 3**, **section 3.3.4**; Catford & Jones, 2019).

Although comparative studies are lacking, the North American prairie appears to be the temperate grassland region most impacted by alien biota. The history of alien species introductions is linked to the arrival and spread of European settlers in the nineteenth century, and subsequent land conversion (Seastedt & Pyšek, 2011), associated with plant introductions having far-reaching consequences such as the conversion of prairies to annual grasslands dominated by Eurasian grasses such as *Bromus tectorum* (downy brome) (Mack, 1989). Intentional introductions have played a key role in this trend (Lehan *et al.*, 2013; Mack & Erneberg, 2002). For the entire United States, the cumulative number of introduced insect, mite (Sailer, 1983), and bird (Temple, 1992) species has grown consistently since the 1800s. In Kansas, a state that falls entirely within the temperate grassland biome, the number of introduced vascular plants found outside of cultivation has been steadily increasing since the late 1800s but has slowed in the last century (Woods *et al.*, 2005). A similar increase-and-decline pattern was reported for rangelands of Washington, Oregon, Idaho, Montana, and Wyoming (testimony of Peter Reich cited in (Mitchell, 2000) and is in line with reports for California (Rejmánek & Randall, 2004) and the United States as a whole (Seebens, Blackburn, *et al.*, 2017).

In South American grasslands, the number of records of alien plants (C. R. Fonseca *et al.*, 2013), invertebrates (De Francesco & Lagiglia, 2007), birds (Zufiaurre *et al.*, 2016) and vertebrates are still increasing. However, formal trend analyses are lacking as are comprehensive reviews or summary data.

Review data on trends are missing for the Eurasian steppe biome. Although new plant species continue to colonize even highly continental Asia (Urgamal *et al.*, 2014), they remain mainly confined to ruderal and otherwise disturbed habitats, while frequency and abundance in natural grasslands remains low. For the extensive grassland regions of Mongolia and China, an increase towards a higher share of C4 plants in the otherwise C3-dominated vegetation has been described (Wittmer *et al.*, 2010). This is, however, attributed to a higher share of native species (*Cleistogenes* spp. and Amaranthaceae weeds) and may partly be triggered by warmer climate. In the middle of the last century, almost all introduced plants in Kazakhstan were either cultivated or confined to ruderal plants, with none recorded as colonizing temperate steppe grasslands (Pavlov, 1956). Compared to other continents, the trend in continental Asia might indicate a lower introduction pressure, harsher climate conditions, or time lag compared to temperate grasslands in other continents.

Status

The total number of organisms introduced to temperate grasslands worldwide has never been assessed thoroughly. A comparison of the proportion of alien species among all species across habitats revealed that temperate grasslands exhibit intermediate levels of invasions with lower proportions than urban or agricultural habitats but higher proportions than wetlands or planted forests (Catford & Jones, 2019). In states that lie entirely within the Great Plains of the United States (Kansas, Nebraska, North Dakota, Oklahoma, South Dakota), 790 alien vascular taxa (14.6 per cent of the flora) are found outside of cultivation, with forbs and herbs comprising the largest group (553 taxa, 70 per cent of the alien flora) (data extracted from the PLANTS Database; USDA, NRCS, 2021). Introduced plant species have become so common in the prairies that grasslands lacking any alien species are rare (S. DeKeyser *et al.*, 2010; Larson *et al.*, 2001). Examples of invasive alien species include perennial C3 (e.g., *Bromus inermis* (awnless brome), *Poa angustifolia* (Kentucky bluegrass); E. S. DeKeyser *et al.*, 2015; Otfinowski *et al.*, 2007) and C4 (*Bothriochloa ischaemum* (yellow bluestem), *Dichanthium sericeum* (silky bluegrass); Mittelhauser *et al.*, 2011; Simmons *et al.*, 2007) grasses introduced as forage grasses, as well as annual grasses (e.g., *Bromus tectorum* (downy brome); Ashton *et al.*, 2016) and biennial and perennial forbs (e.g., *Centaurea stoebe* subsp. *australis* (spotted knapweed), *Euphorbia virgata* (leafy spurge); LeJeune & Seastedt, 2001; Dunn, 1985). Although the rate of introduction appears to have slowed in North American temperate grasslands, the regional expansion and range infilling of already introduced alien species is ongoing (e.g., *Ventenata dubia* (North Africa grass); Wallace *et al.*, 2015).

In the central Great Plains, 14 alien earthworm species occur in the wild (J. W. Reynolds, 2016). Furthermore, *Sus scrofa* (feral pig) – descendants from stock introduced from Europe – have become invasive in the southern and northern Great Plains (Brook & van Beest, 2014; Reeves *et al.*, 2021). *Equus caballus* (horse) have escaped and colonized some areas of Australia and the Great Plains, though are highly restricted in their current range for the latter (Nimmo & Miller, 2007; Reeves *et al.*, 2021). Although trees are scarce in the prairie, some invasive alien species, such as *Agrilus planipennis* (emerald ash borer; insect), *Adelges piceae* (balsam woolly adelgid; insect), and *Ophiostoma* species (Dutch elm disease; fungi; Reeves *et al.*, 2021) can damage trees that grow locally.

In South America, around 350 alien plant species have been recorded for the Pampa regions, of which ca. 50 occur in natural and semi-natural grasslands (C. R. Fonseca *et al.*, 2013). In Brazil, the Pampa region had the highest proportion of established alien species relative to total richness and compared to other natural regions (114 alien established alien species out of 1,685 species in total; Zenni, 2015). Invasive alien species are particularly common in the Pampas of Argentina, but also are abundant and problematic in other temperate grasslands of South America. Pampas are subject to invasion by alien shrubs from Eurasia (Mazía *et al.*, 2010; Zalba & Amodeo, 2015) as well as by herbaceous alien species (Dresseno *et al.*, 2018; Hierro *et al.*, 2011). Similar to North America, the latter include alien species that have been introduced as pasture grasses, especially from Africa (*Eragrostis curvula* (weeping lovegrass), *Eragrostis lehmanniana* (Lehmann lovegrass), *Panicum coloratum* (klein grass; D. G. Williams & Baruch, 2000)), and herbs (Tognetti & Chaneton, 2012). Introduced alien pine species have been planted on a large scale in the high-altitudinal temperate grasslands of the Páramo and are showing signs of escape and spread (Hofstede *et al.*, 2002; van Wesenbeeck *et al.*, 2003).

In contrast, numbers of alien species are low in the harsh continental grassland regions of Asia. Several of the most important alien grasses in North American prairies originate from steppes and related grasslands (*Agropyron cristatum* (crested wheatgrass), *Bromus tectorum* (downy brome)), yet the continental climates of central Eurasia are less invaded. Mongolia, with its ca. 1 million km^2 of steppes, has less than 100 alien plant species (out of ca. 3200;

Urgamal *et al.*, 2014). None of these 100 alien plant species achieved high frequency or dominance in steppes, and the few studies on invasive plants from northern China also refer to heavily disturbed areas, fields or sown grasslands rather than natural steppes (Guan *et al.*, 2019; Xun *et al.*, 2017). The same holds true for the extensive steppes of Kazakhstan and surrounding environments, while the steppes of Russia and Europe are heavily converted (Kamp *et al.*, 2016; Smelansky & Tishkov, 2012). The remaining steppes of these regions often have altered plant community compositions, but the species are overwhelmingly native to the regions. Alien plants are typically confined to arable fields, and ruderal and disturbed areas (Sukhorukov, 2011; Vakhlamova *et al.*, 2016).

Equus caballus (horse; Zalba & Loydi, 2014) and *Sus scrofa* (feral pig; Caruso *et al.*, 2018) are known to occur in South American grasslands. Several alien bird species have established in Pampas such as *Myiopsitta monachus* (monk parakeet; Bucher & Aramburú, 2014) and *Sturnus vulgaris* (common starling; Zufiaurre *et al.*, 2016). Data on invertebrates are more anecdotal, yet invasions have been documented for *Rumina decollata* (decollate snail; De Francesco & Lagiglia, 2007).

Data and knowledge gaps

Alien plant invasions in temperate grasslands in the Americas are reasonably well documented in the scientific literature. By comparison, the frequency and impact of other alien taxonomic groups, such as earthworms, remain understudied in these regions. Numbers of documented alien species from the steppes of inner Asia are low and it seems likely that records are missing due to low research intensity and that higher numbers could be expected, particularly in countries of low economic growth.

Records on alien animal species are incomplete with only limited reports available on common invasions in Asia. Widespread alien mammals, such as *Mus musculus* (house mouse), are even thought to have large parts of their native range in continental Asia (Appenborn *et al.*, 2021). Baseline data are available for invertebrates and although far from comprehensive.

2.5.2.7 Deserts and xeric shrublands

Deserts and xeric shrublands correspond, in general, to regions with low population densities and several are located in countries with low per capita gross domestic product. Due to their harsh climate, few alien plants have been able to establish in these habitats (Kalusová *et al.*, 2017). As such, they are expected to harbour fewer alien and invasive alien species than other biomes (Dawson *et al.*, 2017). On the other hand, the harsh abiotic conditions sometimes motivated the introduction of alien species capable of surviving in such habitats to ameliorate human livelihood.

Trends

Comparing rates of alien plant species accumulation, accounting for area, the accumulation of alien plants appears to be slower in deserts and xeric habitats than in colder temperate and Mediterranean regions (Pyšek, Pergl, *et al.*, 2017). Although these habitats used to be considered relatively resistant to alien plant invasion, the recent spread of alien species has been observed (Sandquist, 2014). In Chinese desert areas, the number of new invasive alien species is increasing (Eminniyaz *et al.*, 2017) although this finding could also be explained by changing recording intensities. *Prosopis juliflora* (mesquite) was introduced to many desert regions starting in the 1850s and is now a widespread invader in all regions except Europe and Central Asia (Patnaik *et al.*, 2017). *Cenchrus ciliaris* (buffel grass) was widely introduced in the early 1900s for forage and pasture and now invades large areas in Australia and Americas where it increases wildfire frequency and intensity (V. M. Marshall *et al.*, 2012). *Camelus dromedarius* (dromedary camel) were introduced in the 1800s in Australia to assist transportation across deserts and later escaped and spread (Crowley, 2014).

The number and accumulation of emerging alien species worldwide is expected to continue to increase for most taxonomic groups and continents, though possibly more slowly in deserts and xeric shrubland compared to other biomes. Other studies predict that deserts will be unsuitable for invasive alien species by 2100 (Bellard, Thuiller, *et al.*, 2013). Trade and transport in the subtropics (a zonobiome overlapping much of deserts and xeric shrublands) is expected to be the main driver facilitating biological invasions (Essl *et al.*, 2020), although these areas have comparatively less trade and transport than other more populated regions (subtropics cover approximately 25 per cent of the terrestrial surface of the planet but only have 8 per cent of world population).

Status

Global analyses (Dawson *et al.*, 2017; Turbelin *et al.*, 2017) show some tendency for lower richness of established alien species in deserts and xeric shrublands than in temperate and Mediterranean biomes, but with some variation among regions. The Palearctic deserts in Central Asia and north Africa and the Sahara and Afrotropic deserts south of the Sahara in Africa and the southern fringe of the Arabian Peninsula (with some exceptions, e.g., Southern Africa) show relatively low numbers of alien and invasive alien species. The Australasian deserts, the Nearctic deserts in North America, the Neotropical deserts in South America and the Indo-Malay deserts south of the Himalayas tend to harbour higher numbers of established alien species, although generally much lower compared to Temperate and Mediterranean regions (Dawson *et al.*, 2017; Turbelin *et al.*, 2017).

The different taxonomic groups show some differences both in numbers of established alien species (many more plants than animals) and regionally. The number of established alien plants is generally lower in desert areas than in temperate and Mediterranean climates (e.g., 119 alien plants in the Nama karoo and 75 in the Succulent karoo, both in South Africa (B. W. van Wilgen & Wilson, 2018) and 73–83 alien plants in several parks of the North American Mojave Desert (Abella *et al.*, 2015). In the desert region of Egypt only 17 alien species were reported (Shaltout *et al.*, 2016). Following European settlement of Australia, numerous alien plant species were intentionally introduced for use in crops, pastures, gardens, and horticulture, and others arrived unintentionally. Many subsequently escaped into natural environments and are now considered as "weeds". Of the 54 alien plant species of natural environments of arid and semi-arid Australia that are considered here, 27 were apparently unintentionally introduced, 20 were intentionally introduced, and 7 were probably introduced both unintentionally and intentionally. Livestock, including camels and their harness, and contaminated seed and hay were the most common vectors for unintentional introduction (Crowley, 2014; Friedel, 2020).

Established alien birds are absent or present in only low number in most desert and xeric habitats of the world, with a few exceptions in North American and Southern African deserts (B. W. van Wilgen & Wilson, 2018), possibly because there were few attempts to intentionally introduce alien birds in arid regions (E. E. Dyer, Cassey, *et al.*, 2017). The number of established alien freshwater fishes is similar in both Australian and African deserts but tends to be higher in American and Asian deserts; their occurrences are associated with oases, as is the case of at least four alien freshwater fish species found in the largest oasis in the Mojave Desert (Ash Meadows), in North America (Scoppettone *et al.*, 2005). The number of alien reptiles and amphibians introduced to deserts and xeric habitats is low (mostly below four) compared to other biomes. Regional comparisons indicate lower numbers for Palearctic deserts in Eurasia north of the Himalayas and in north Africa as well as for the Sahara, especially for amphibians (Capinha *et al.*, 2017) than other deserts. In Southern African deserts, none or only one alien species has been reported (B. W. van Wilgen & Wilson, 2018). In a survey of eleven oases in the desert regions of Morocco, five alien ant species have been recorded spreading across seven oases (A. Taheri *et al.*, 2021). Information about alien spiders is missing in many regions; in African deserts there are almost no alien spider species or they are not studied, but in Australian and American deserts, the numbers do not differ much from other biomes (Dawson *et al.*, 2017). For other animal groups, fungi, and microorganisms, little information was available except for the presence of *Batrachochytrium dendrobatidis* (chytrid fungus) associated with declines and extinctions of amphibians worldwide, in oasis of the Baja California Sur Desert, in Mexico (Luja *et al.*, 2012).

Data and knowledge gaps

Deserts and xeric shrublands are less well-studied relative to other biomes (e.g., Crystal-Ornelas & Lockwood, 2020; Florencio *et al.*, 2019). Global studies provide information on the status of alien species in the different desert and xeric shrubland regions, but information on temporal trends is often incomplete or even absent for most deserts. Most available studies focus on plants and animals (but not arthropods) and there were almost no studies on fungi and microorganisms (Pyšek, Hulme, *et al.*, 2020). There is more information for the deserts of North America, but for other less well-surveyed regions, for example Africa (except South Africa) and Asia, information is scarce and limited to few species. The lack of information is particularly concerning because arid areas and desertification may be expected to increase in the future.

2.5.2.8 Cryosphere

Trends

The cryosphere has been less affected by alien species compared to other regions. The low number of reported alien species from the cryosphere have multiple reasons: The cryosphere is difficult to access, anthropogenic pressures have been low (Bennett *et al.*, 2015; Chan *et al.*, 2019; Galera *et al.*, 2018; McGeoch *et al.*, 2015; Ruiz & Hewitt, 2009; Vermeij & Roopnarine, 2008) and inhospitable environments (e.g., low nutrient soils, freezing temperatures, high UV levels) do not favour establishment of alien species. Although the Arctic and Antarctica differ, climate change and increased human activities (tourism and research) are enhancing introductions in both regions (**Chapter 3**, **Box 3.4**; Bartlett *et al.*, 2020; Bender *et al.*, 2016; Cárdenas *et al.*, 2020; Chan *et al.*, 2019; Chown *et al.*, 2012; Chwedorzewska *et al.*, 2015; Duffy *et al.*, 2017; Frenot *et al.*, 2005; K. A. Hughes, Cowan, *et al.*, 2015; K. A. Hughes, Pertierra, *et al.*, 2015; Huiskes *et al.*, 2014; McCarthy *et al.*, 2019; McGeoch *et al.*, 2015; Miller & Ruiz, 2014; Ricciardi *et al.*, 2017; Wasowicz *et al.*, 2020). Plants (seeds, fragments and other propagules) and invertebrates (e.g., springtails) are introduced on clothing and personal equipment of tourists, ships, and aircraft personnel, as well as associated with packing materials (Chown *et al.*, 2012; Huiskes *et al.*, 2014), vehicles (K. A. Hughes *et al.*, 2010), and fresh food imports (K. A. Hughes *et al.*, 2011). In ten years of surveillance (2007-2017; **Glossary**) at the Scott Base in the Ross Sea region of continental Antarctica, 68 invertebrate species (15 alien within the broader Antarctic region) were intercepted on food (60 per cent), clothing and equipment (11 per cent), aircraft and cargo (11 per cent), and packaging material (11 per cent) (Newman *et al.*, 2018). During 2007-2008 in Antarctica, over 20 alien lichens and fungi were intercepted in packaging, foodstuffs, and timber (Osyczka, 2010; Osyczka *et al.*, 2012). Similarly, 1,019 seeds were found under the footwear of 259 travellers to Svalbard during summer 2008 alone (Ware *et al.*, 2012), while the seeds of eight alien plant species were reported in the topsoil of Fildes Peninsula, King George Island (Antarctica), in areas intensively frequented by humans (Fuentes-Lillo *et al.*, 2017).

In the Arctic marine environment, the rate of reported alien species rose sharply from the end of 1990 concomitantly with increased research efforts in the region. Biofouling on commercial ships is not considered an important pathway for marine alien species for the cryosphere due to the low rate of species survival (but see Chan *et al.*, 2019), while biofouling on other vessel types (e.g., leisure crafts, fishing vessels, floating platforms) could become relevant in the future for the recent increase in tourism, fisheries, and oil and gas development in the Arctic (Chan *et al.*, 2019). Species were mainly introduced by ballast water followed by natural spread from neighbouring areas where the species were first introduced, and by aquaculture activities (e.g., *Paralithodes camtschaticus* (red king crab); Chan *et al.*, 2019; Orlov & Ivanov, 1978). Similarly, in the Antarctic marine environment, species were likely introduced by vessels (three by hull fouling, one by ballast water), with the first recorded alien species (a bryozoan) dating back to 1960, followed in 1986 by a crab, and in 1996 by a tunicate and a hydroid; the most recent introduction (a mollusc) was recorded in 2019, although it is likely that this species has subsequently gone extinct (Cárdenas *et al.*, 2020; McCarthy *et al.*, 2019). It is important to note that there is no evidence that any of these species (bryozoan, crab, tunicate, hydroid) are established in the Antarctic (McCarthy *et al.*, 2019). Terrestrial alien plants in the cryosphere consist of predominantly herbaceous species, mostly introduced inadvertently in association with soils or imported fodder for domestic animals (Chwedorzewska *et al.*, 2015; Frenot *et al.*, 2005; Wasowicz *et al.*, 2020). In the Arctic, there are some records of alien neophyte plants reported at the end of the nineteenth century, but their number increased in the 1950s and 1970s with species mostly introduced by seed contamination and transport on vehicles (Wasowicz *et al.*, 2020). In continental Antarctica, few alien plants have been introduced since the 1950s (e.g., *Poa pratensis* (smooth meadow-grass) was introduced unintentionally during tree transplantation experiments in the 1950s and was eradicated in 2015 (Pertierra *et al.*, 2017).

A comprehensive review on alien invertebrates is missing for the Arctic, but detailed data are reported for the Svalbard archipelago (e.g., Wieczorek & Chłond, 2019), with 32 alien invertebrates recorded since 1928 with an increase after 1980s, mostly due to soil importation (Coulson, 2015). In continental Antarctica, alien invertebrates, such as the springtail *Hypogastrura viatica* (springtail), were reported from the 1940s onwards (Hack, 1949; K. A. Hughes, Pertierra, *et al.*, 2015). In terms of alien vertebrates in the Arctic, four fishes (salmonids) were translocated from North America to Scandinavia and Russia for fisheries and aquaculture since the end of 1800 (Lento *et al.*, 2019), some mammals were intentionally farmed (e.g., *Mustela vison* (American mink) from the 1920s), while others unintentionally arrived in the 1960s (e.g., *Microtus levis* (sibling vole) in Svalbard; Sandvik, Dolmen, *et al.*, 2019). In the Antarctic region, alien vertebrates have been reported only for sub-Antarctic islands where they can survive (conditions in the Antarctica itself are probably too extreme unless the species can live synanthropically): some mammals (i.e., rats and mice) were unintentionally introduced since the eighteenth century, others (such as ungulates, cats, rabbits, salmonids) were intentionally introduced beginning in the 1950s (Frenot *et al.*, 2005; Lecomte *et al.*, 2013).

The number of alien species in the cryosphere is expected to increase in the future due to climate change and human pressure (**Chapter 3**, **sections 3.2.2** and **3.3.4**), but reported numbers are also expected to be higher due to the greater research effort, as noted by the growing number of publications on this area (Chan *et al.*, 2019; Chwedorzewska *et al.*, 2020; Duffy *et al.*, 2017; K. A. Hughes & Pertierra, 2016; Ricciardi *et al.*, 2017). A recent exercise of horizon scanning for future potentially invasive alien species in the Antarctic Peninsula underlined the main threat posed by marine invertebrates that can be unintentionally transported in ballast waters and on ship hulls (K. A. Hughes *et al.*, 2020; McCarthy *et al.*, 2019). The threat could be even greater considering the cruise ship volume from the Northern Hemisphere to Antarctica that may increase the probability of introducing species able to survive cold environments (Chwedorzewska *et al.*, 2020).

Status

In the Arctic, 34 marine alien species have been reported, mostly crustaceans, seaweed, fish, and molluscs (Chan *et al.*, 2019). Many more alien species are expected to arrive in the future, with Hudson Bay, Northern Grand Banks, Labrador, Chukchi, Eastern Bering Seas, and Barents and White Seas considered to be the most vulnerable areas (Goldsmit *et al.*, 2020). 341 alien plants (188 established and 11 invasive) are reported, and their numbers are expected to increase due to a warmer climate (Wasowicz *et al.*, 2020). The Svalbard archipelago is one of the most studied Arctic areas for biodiversity and alien species: 98 alien and 5 established alien species are reported (Sandvik, Dolmen, *et al.*, 2019), mostly coming from mainland Norway.

Most alien species cannot survive in Antarctic continental conditions, but several have been able to adapt to new territories by remaining in the vicinity of human settlements (i.e., research stations), where they can reproduce in more favourable conditions (K. A. Hughes *et al.*, 2010; McGeoch *et al.*, 2015). Up to now, only five marine alien invertebrate species have been found (plus one cryptogenic seaweed species) with free-living specimens but not established populations (Cárdenas *et al.*, 2020; McCarthy *et al.*, 2019). This low number of recorded marine alien species in Antarctica could be due to very harsh environmental conditions (harsher than the Arctic), incomplete assessment of local biodiversity, and limited sampling efforts (McCarthy *et al.*, 2019). For terrestrial species in the continental Antarctic (sub-Antarctic islands excluded), there are 15 known alien species – *Poa annua* (smooth meadow-grass) and 14 invertebrates (7 Collembola, 4 Arachnida, 2 Insecta Diptera, 1 Annelida), most of which are found in the Antarctic Peninsula region (Baird *et al.*, 2019; Enríquez *et al.*, 2019; K. A. Hughes *et al.*, 2020; K. A. Hughes, Pertierra, *et al.*, 2015). This could be due to several factors. This Antarctic Peninsula is the area closest to another continent (South America), it is the least climatically extreme region of Antarctica (and has also experienced a rapid rise in temperatures since the 1950s due to climate change), and it has the largest concentration of human activity (due to research teams and tourism) resulting in a relatively high propagule pressure (K. A. Hughes *et al.*, 2020). On the sub-Antarctic islands, which circle the continent, at least 108 alien plants, 72 terrestrial invertebrates, 16 vertebrates are reported (Frenot *et al.*, 2005).

Data and knowledge gaps

Overall, the trends and status of alien species in the cryosphere could be better documented, even if the number of studies on this biome rapidly increased in the last years (Chwedorzewska *et al.*, 2020). However, baseline biodiversity knowledge is poor and suitable taxonomic expertise is often lacking, making it difficult to identify alien species, particularly invertebrates and aquatic species (K. A. Hughes & Convey, 2012). For example, freshwater biodiversity is low in continental Antarctica, generally dominated by cyanobacteria, cyanophytes, bacteria, yeasts, rotifers, nematodes and diatoms; as yet, there are no reports of established alien species, but taxonomic specialists of freshwater and terrestrial Antarctic biota are rare (K. A. Hughes *et al.*, 2020; K. A. Hughes & Convey, 2012).

2.5.3 Trends and status of alien and invasive alien species in freshwater units of analysis

2.5.3.1 Wetlands – peatlands, mires, bogs

Trends

Contrary to other freshwater wetlands, peatlands, mires, and bogs have generally been considered more resistant and resilient to biological invasions due to their extreme environments (such as low nutrients and oxygen, harsh climate in high mountains or salinity) and absence of anthropogenic pressure for many years (Chytrý *et al.*, 2008; Parish *et al.*, 2008; Zefferman *et al.*, 2015). However, landscape transformation, due to peatland drainage for agriculture, peat extraction, deforestation, road construction, and increased international trade since the nineteenth century, is facilitating an increase of alien species in these ecosystems (Miletti *et al.*, 2005; Parish *et al.*, 2008; Rebelo *et al.*, 2018; Catford *et al.*, 2017; Pellerin & Lavoie, 2000; Tousignant *et al.*, 2010). Indeed, many peatlands have been drained for agriculture or mined for peat, which has greatly altered their plant communities. For example, 98 per cent of the fens of the state of Ohio, United States, have been destroyed, and invasion by alien species is an ongoing concern in many remaining fens (Andreas, 1989). In Asia, increased numbers of aquatic invasive alien plants are low (0-5 species) in five countries in the region during a period of 7-18 years (Banerjee *et al.*, 2021; Government of Myanmar,

Box 2 8 **Rapid rise of alien fishes in the Amazon, the world's most biodiverse freshwater region.**

The Amazon region contains the world's richest native diversity of fishes (Toussaint *et al.*, 2016). The extent to which this global centre of endemism has been invaded by alien species has been largely overlooked. A recent study involving 35 regional experts has documented 41 species and 17 families of alien fishes in the region, based on records that extend as far back as 1939 (Doria *et al.*, 2021). Most (75 per cent) of these records were observed since the year 2000, during which time there has been a distinct increase in the accumulated number of alien species with no sign of saturation. This is in contrast to the classical view that biodiverse regions are resistant to invasion. More than half of these alien species are omnivores or carnivores, and are distributed for use in aquaculture or the aquarium trade. Intensive fish farming, in particular, is deemed to be a major burgeoning contributor to species introductions in the region (e.g., Doria *et al.*, 2020).

2005; Islam *et al.*, 2003; Khuroo *et al.*, 2012; Mukul *et al.*, 2020; Pallewatta *et al.*, 2003; Shrestha & Shrestha, 2021; D. T. Tan *et al.*, 2012; Tiwari *et al.*, 2005; Wijesundara, 2010). A lack of baseline data from most countries impedes comprehensive analysis. Increasing anthropogenic threats posed to non-permanent wetlands, including climate change, will likely accelerate the establishment of new alien species (Catford *et al.*, 2013).

Status

Some studies confirm the lower vulnerability of peatlands to biological invasions, with few or even no alien species reported for these areas (Chytrý *et al.*, 2008; Lambdon *et al.*, 2008; Rejmánek *et al.*, 2013; Zedler & Kercher, 2004). For example, in Europe almost 10 per cent of all alien plants occur in peatlands (Lambdon *et al.*, 2008) with frequency of plants introduced after 1500 spanning from 0 in Catalonia and Czech Republic to 0.2 per cent in the United Kingdom (Chytrý *et al.*, 2008). An assessment of Natura 2000 areas in Poland (Perzanowska *et al.*, 2019) showed that the majority of bogs, mires, and fens host a low number of alien species (maximum 10 species), occurring at low frequency. Other studies underline the increasing effect of the anthropogenic pressures on peatlands and the subsequent higher occurrence of alien species (e.g., Jukonienė *et al.*, 2015).

In contrast to peatlands and bogs, riparian habitats are among the most invaded habitats (Catford & Kyle, 2016; Vilà *et al.*, 2007). A study comparing numbers of established species in European habitats (Pyšek, Bacher, *et al.*, 2010) showed that riparian and aquatic habitats are most heavily colonized by alien mammals and herptiles; the latter group is also reaching high species densities in mires. The highest densities of alien bird species are found in aquatic and cultivated habitats. Overall, riparian habitats appear highly invaded by all groups of animal taxa except insects. For plants, alien species numbers from riparian habitats were almost as high as for urban habitats (Pyšek, Bacher, *et al.*, 2010).

Across Asia, the number of invasive alien plants in non-permanent freshwater ecosystems range from 5-13 species in 13 countries (Banerjee *et al.*, 2021; Kurniawan & Paramita, 2020; Mukul *et al.*, 2020; Qureshi *et al.*, 2014; Shrestha & Shrestha, 2021; Sujanapal & Sankaran, 2016; Weber *et al.*, 2008; Wijesundara, 2010; H. Xu *et al.*, 2012). The most dominant species in the region are *Pontederia crassipes* (water hyacinth, recorded in 17 countries of the 19 countries for which data are available), *Pistia stratiotes* (water lettuce, 17), *Salvinia × molesta* (kariba weed, 12), *Mimosa pigra* (giant sensitive plant, 11), and *Alternanthera philoxeroides* (alligator weed, 10).[5] Some of the new additions to the region include *Cabomba caroliniana* (Carolina fanwort) and Typha angustifolia (lesser bulrush). In Kolkheti Lowland (Georgia), 423 alien plants are reported, 308 of which are present in peatland areas: the introduction of these species was favoured by the increased transformation and anthropization of the areas in the nineteenth century (Parish *et al.*, 2008). Wagner *et al.* (2017) found that, among the 83,396 plots of woodland habitats in Europe, broadleaved bog woodlands on acid peat have the second highest mean relative alien species richness per plot (2.2 per cent), probably due to a higher degree of human disturbance (e.g., peat extraction) and the invasiveness (**Chapter 1**, **section 1.4.3**) of some alien species like *Prunus serotina* (black cherry).

Drainage can favour the accessibility of these areas for tourists, facilitating the unintentional introductions of alien species (Parish *et al.*, 2008): in 2018 *Drosera rotundifolia* (common sundew) was found in a peat bog in Nahuel Huapi National Park (Argentina) and its introduction seems related to tourists visiting the area (Vidal-Russell *et al.*, 2019). Other disturbances can promote alien species introduction and spread: in the montane bogs of Haleakala National Park, Hawaii, undisturbed bogs were less invaded, while bogs with feral alien pigs showed an increase in invasive alien plants (Loope *et al.*, 1992). A similar result was found in other areas: in a *Sphagnum*-dominated peatland in the Central Andes of Colombia, increased nutrient additions and physical disturbance due to agricultural activities led to the widespread occurrence of *Cenchrus clandestinus* (Kikuyu grass; Urbina & Benavides, 2015); in a New Zealand bog modified by the surrounding agricultural activities, a higher occurrence of alien invertebrates has been reported compared to undisturbed bogs (Watts *et al.*, 2020). Finally, in some cases, natural and prescribed fires can favour biological invasions in these ecosystems. At Kaituna Wetland, Bay of Plenty (New Zealand), fire disturbance promoted more alien species (Christensen *et al.*, 2019): after four years, the authors found 14 alien vascular species and 10 native species in burnt plots vs 10 alien species and 18 native species in unburnt plots. A similar situation is reported for the United Kingdom where in burnt plots the invasive alien moss *Campylopus introflexus* (heath star moss) was more abundant and present than the native cotton grass *Eriophorum vaginatum* (hare-tail cotton-grass; Noble *et al.*, 2017).

Data and knowledge gaps

There is a lack of comprehensive and in-depth studies on alien species in peatlands across different continents and involving all taxa. The literature mostly presents scattered specific studies, focused on Europe and North America, which are biased towards plants. Information about the temporal trends of alien species in peatlands, bogs and mires, and their status are also mostly missing.

5. Data extracted from the GISDP (http://www.iucngisd.org/gisd/), GRIIS (https://doi.org/10.5281/zenodo.6348164) and ASEAN (https://asean.org/)

2.5.3.2 Inland surface waters and water bodies/freshwater

Trends

The number of alien species in freshwater has been reported to increase all over the world (Cowie, 1998; Hussner *et al.*, 2010; O'Flynn *et al.*, 2014; Ricciardi, 2001; Roll *et al.*, 2009). The trends in rising alien species numbers are very consistent across all taxonomic groups such as aquatic invertebrates (Mangiante *et al.*, 2018; Muñoz-Mas & García-Berthou, 2020; Rabitsch & Nehring, 2017; Roll *et al.*, 2009), vertebrates (A. B. Kumar, 2000; Muñoz-Mas & García-Berthou, 2020) and plants (Hussner *et al.*, 2010; Mangiante *et al.*, 2018; Rabitsch & Nehring, 2017), across habitats such as lakes (Ricciardi, 2001) and rivers (M. C. Jackson & Grey, 2013; Rabitsch & Nehring, 2017) and across continents such as Europe (M. C. Jackson & Grey, 2013; Rabitsch & Nehring, 2017), North America (Mangiante *et al.*, 2018; Ricciardi, 2001), and Asia (Roll *et al.*, 2009). Comprehensive studies for Africa, Australasia, and South America (**Boxes 2.8** and **2.9**) are mostly lacking, but global studies and studies of individual taxonomic groups suggest similar increasing trends (Madzivanzira *et al.*, 2021). In many cases, increases in freshwater alien species numbers accelerated after 1950 (Chambers *et al.*, 1999; Hussner *et al.*, 2010; Mangiante *et al.*, 2018; Mills *et al.*, 1993; Muñoz-Mas & García-Berthou, 2020; Roll *et al.*, 2009), while other studies show consistent increases since 1900 (Rabitsch & Nehring, 2017) or even 1800 (Ricciardi, 2001). The observed acceleration may, however, also result from increased sampling intensity and greater awareness in more recent years (Belmaker *et al.*, 2009; C. J. Costello & Solow, 2003).

Numbers of alien freshwater vertebrates seem to have been increasing for longer compared to invertebrates, although this may also be a consequence of varying sampling intensity and better taxonomic and ecological knowledge (Muñoz-Mas & García-Berthou, 2020). The number of alien insects in freshwaters is comparatively low even though aquatic insects are frequent in native faunas (Fenoglio *et al.*, 2016; Guareschi *et al.*, 2013; Muñoz-Mas & García-Berthou, 2020). This has been attributed to a combination of factors including low economic impact and low probability of transport and survival of alien aquatic insects (Fenoglio *et al.*, 2016). Furthermore, not only has the number of freshwater alien species consistently increased, but the rates of new records over time also rose continuously (M. C. Jackson & Grey, 2013; Leuven *et al.*, 2009; Muñoz-Mas & García-Berthou, 2020; Ricciardi, 2001). Declines in new records of alien species have been observed in a few studies recently (i.e., after 2005), but these declines are likely due to lags in detection and reporting of new alien species (Mangiante *et al.*, 2018; Muñoz-Mas & García-Berthou, 2020; Seebens, Blackburn, *et al.*, 2017). Increases in either species numbers or rates of new records have been associated with increasing import volumes (Cowie, 1998; M. C. Jackson & Grey, 2013; Ricciardi, 2001; Seebens, Essl, *et al.*, 2017), human population size (M. C. Jackson & Grey, 2013), and tourism (Cowie, 1998). Similar increases are reported for alien plants as shown by the Joint Nature Conservation Committee River Macrophytes Database that contains records from standardized vegetation surveys of rivers from across the United Kingdom. Surveys focus on rivers with existing or potential conservation value, and almost 4500 surveys have been undertaken since 1977. River sites were surveyed both pre- and post-1990. Results showed a 31 per cent increase in the presence of invasive alien plant species across two survey periods in the United Kingdom (Pattison *et al.*, 2017).

Status

Although probably due in large part to a knowledge bias, biological invasions in aquatic systems represent only a small fraction of all invasions; for example, of the 2,033 alien species recorded in South Africa, only 191 are aquatic; of these, most are freshwater invasive alien species (Skowno *et al.*, 2019). Global maps of the distribution of alien species exist for fishes (Dawson *et al.*, 2017; Leprieur *et al.*, 2008) and amphibians (Capinha *et al.*, 2017). In both cases, consistently high numbers of alien freshwater species have been reported for Europe and North America, including Hawaii, while hotspots of alien freshwater fishes have also been found in South-East Asia, Central Asia and mesoamerica (e.g., Dawson *et al.*, 2017; Leprieur *et al.*, 2008; **Boxes 2.8** and **2.9**). Leprieur *et al.* (2008) reported occurrences of 9,968 alien fish species in 1,055 river basins worldwide, with up to 95 per cent of present fish species being alien. The global distribution of alien freshwater fishes has been attributed to high per capita gross domestic product and high human population density (**Chapter 3**, **sections 3.2.2** and **3.2.3.6**; Dawson *et al.*, 2017; Leprieur *et al.*, 2008), but also high per centages of urban areas and basin areas (Leprieur *et al.*, 2008). Many alien freshwater species have been intentionally released (A. B. Kumar, 2000; Muñoz-Mas & García-Berthou, 2020; Strayer, 2010) through, for instance, recreational fishing (Davis & Darling, 2017). Introduced fish species often represent large-bodied species (predators and herbivores) (Blanchet *et al.*, 2010), which may alter food web structures with consequences for the whole food web (Cucherousset *et al.*, 2012). Capinha *et al.* (2017) report alien populations for 78 amphibian species, but not all might be classified as freshwater species. Significantly more alien freshwater amphibians have been found in islands compared to mainlands (Capinha *et al.*, 2017). An important pathway for introduction is the construction of inland canals which are responsible for a large number of freshwater alien species such as invertebrates and fish (Faulkner *et al.*, 2020; Galil *et al.*, 2007; Katsanevakis *et al.*, 2013; Schöll, 2007). Among alien freshwater invertebrates, most studies are available

Box 2 9 North American Great Lakes: An assessment of trends of alien species.

The biological invasion history of a region can reveal the changing influence of transport vectors and management actions over time. The North American Great Lakes basin is the world's most invaded freshwater ecosystem (Pagnucco *et al.*, 2015; Ricciardi, 2006). Numbers and taxonomic composition of established alien species discovered in the basin during different time periods are correlated to changes in vector and pathway activities, such as fish stocking, canal development, and transoceanic shipping (Ricciardi, 2006). Thus, the biological invasion history of the basin is punctuated by major phases distinguished by a predominance of particular taxonomic and functional groups as well as taxa from particular donor regions. During periods of fish stocking, for example, fishes and fish pathogens comprised many of the alien species discovered. Similarly, following the transition from solid ballast to ballast water in ships during the early twentieth century, alien species of phytoplankton and zooplankton were discovered more frequently (Mills *et al.*, 1993). The opening of the St. Lawrence Seaway in 1959 marked a period in which ballast water discharge became the dominant vector of invasion. A more recent phase in the history of the basin is distinguished by a mass invasion of Ponto-Caspian species (including *Dreissena polymorpha* (zebra mussel), *Dreissena rostriformis bugensis* (quagga mussel), *Neogobius melanostomus* (round goby), *Cercopagis pengoi* (fishhook waterflea), and several others) and euryhaline invertebrate taxa with resting eggs that can survive transport in ballast tank sediments (Pagnucco *et al.*, 2015; Ricciardi, 2006; Ricciardi & MacIsaac, 2000). Between 1959 and 2006, inclusive, the average rate of discovery of newly established alien species in the basin was 1.69 per year, or one new alien species every 7 months **(Figure 2.35)**. The majority (65 per cent) of these introductions are attributable to ballast water shipping, primarily from European donor regions. However, since 2006, the overall rate of invasion has been reduced, declining by 85 per cent to its lowest level in two centuries (Ricciardi & MacIsaac, 2022) with very few invasions attributable to shipping. This abrupt shift in invasion risk follows the implementation of ballast water regulations by Canada and the United States in 2006 and 2008, respectively.

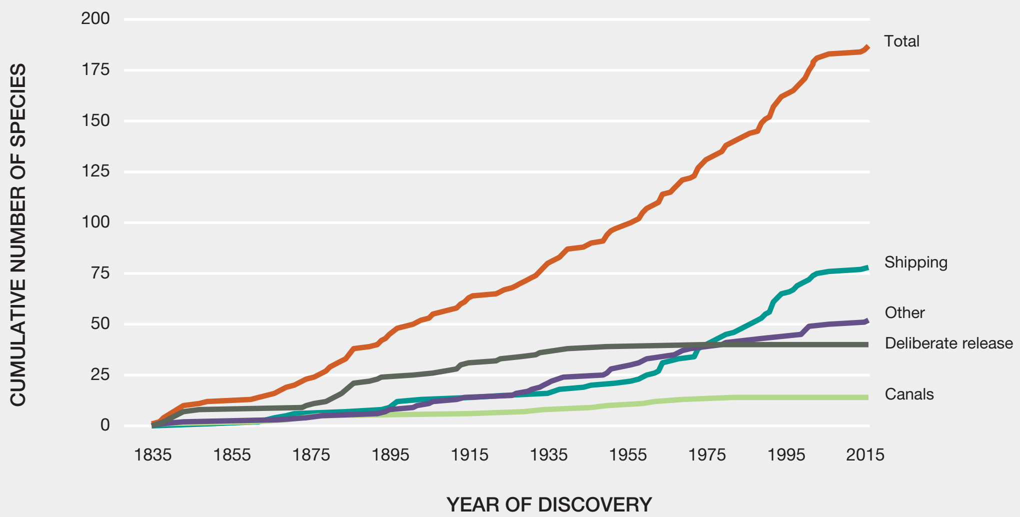

Figure 2 35 Cumulative numbers of alien species in the North American Great Lakes basin.

The total number of alien species is shown in the top most line. Other trend lines show accumulations of species whose introductions are attributable to various vectors, including shipping (ballast water, solid ballast, and hull fouling), canals, deliberate release (e.g., intentional stocking of fishes), and other vectors (e.g., bait, aquarium, and unintentional releases). Data sources: Mills *et al.*, 1993; NOAA, 2021; Ricciardi, 2006.

for freshwater crustaceans and molluscs (Cianfanelli *et al.*, 2016; Cuthbert *et al.*, 2020; Lodge *et al.*, 2012), but no single study exists that shows the global distribution of alien freshwater invertebrates. Compared to aquatic alien animals, aquatic alien plants and algae have been under-investigated. Comprehensive reports on large-scale distributions of aquatic plants are lacking, but global assessments are available for well-investigated individual species such as *Pontederia crassipes* (water hyacinth) (Kriticos & Brunel, 2016), *Azolla filiculoides* (water fern) (Rodríguez-Merino *et al.*, 2019) or *Lemna minuta* (least duckweed) (Ceschin *et al.*, 2018).

Data and knowledge gaps

Inland waters, riparian networks, and channels are very effective corridors for propagules that can easily be dispersed over long distances (Brundu, 2015a; Willby, 2007), but aquatic environments are difficult to monitor and an early detection of a submerged species introduction is seldom possible. No analysis reporting gaps in trends and status of alien species in freshwater systems currently exists, but a comparison of available literature reveals that freshwater systems have been far less investigated than terrestrial and (most likely) marine systems (Seebens, Blackburn, *et al.*, 2017). Among these, the vast majority of studies have been conducted in Europe and North America, while information about the temporal trends in freshwater alien species and their status across continental ranges are largely absent. The only exceptions seem to be fishes and amphibians, for which comprehensive large-scale analyses are available (Capinha *et al.*, 2017; Dawson *et al.*, 2017; Kraus, 2009; Leprieur *et al.*, 2008). However, large information gaps on species occurrences exist among these taxonomic groups, particularly in Asia and Africa (Dawson *et al.*, 2017). Large-scale information is missing for most freshwater invertebrates, including macrophytes and algae. Riparian habitats have been extensively studied for plant invasions (Maskell *et al.*, 2006; D. M. Richardson *et al.*, 2007), but many studies focus on a handful of invasive alien taxa (e.g., Elderd, 2003; Hood & Naiman, 2000; Pyšek & Prach, 1993).

2.5.4 Trends and status of alien and invasive alien species in marine units of analysis

2.5.4.1 Shelf ecosystems (neritic and intertidal/littoral zone)

Trends

The number of marine alien species has been consistently and continuously increasing globally (Bailey *et al.*, 2020) and in individual regions such as in the waters of North America (Cohen & Carlton, 1998; Ruiz, Fofonoff, *et al.*, 2000), Europe (Gollasch, 2006; Katsanevakis *et al.*, 2013; Reise *et al.*, 1998; Zenetos & Galanidi, 2020), Australia (Hewitt *et al.*, 2004), South America (Schwindt *et al.*, 2020; Teixeira & Creed, 2020; Toral-Granda *et al.*, 2017), Africa (Mead *et al.*, 2011; T. B. Robinson *et al.*, 2020) and the Pacific (Carlton & Eldredge, 2009; Coles *et al.*, 1999). Time series of newly reported marine alien species often date back to the early nineteenth century (Carlton *et al.*, 2019; Carlton & Eldredge, 2009; Cohen & Carlton, 1998; Coles *et al.*, 1999; Gollasch, 2006; Hewitt *et al.*, 2004; Mead *et al.*, 2011; Reise *et al.*, 1998; Ruiz, Fofonoff, *et al.*, 2000; Schwindt *et al.*, 2020; Teixeira & Creed, 2020; S. L. Williams & Smith, 2007; Wolff, 2005). Likewise, increases in rates of new alien species records were frequently observed especially in the early twentieth century (Carlton & Eldredge, 2009) or after 1950 (Bailey *et al.*, 2020; Coles *et al.*, 1999; Gollasch, 2006; Hewitt *et al.*, 2004; Mead *et al.*, 2011; Ruiz, Fofonoff, *et al.*, 2000; Schwindt *et al.*, 2020; Teixeira & Creed, 2020; S. L. Williams & Smith, 2007). Wolff (2005) reported an increase in long-distance introduction events after 1950. Increases in marine alien species numbers are not only related to the intensifications of global shipping consistently across studies (i.e., hull fouling and ballast water), aquaculture and cultivation (including stocking and aquarium releases) (Bailey *et al.*, 2020; Coles *et al.*, 1999; Gollasch, 2006; Hewitt *et al.*, 2004; Katsanevakis *et al.*, 2013; Reise *et al.*, 1998), but also increased tourism (Toral-Granda *et al.*, 2017), and natural dispersal from neighbouring alien populations (Gollasch, 2006; Wolff, 2005). Rising shipping activity during both world wars is associated with new marine alien species introductions at naval bases (Coles *et al.*, 1999). Another major pathway was the opening of new shipping canals such as the Panama Canal, the Suez Canal, and the St. Lawrence River (Galil *et al.*, 2007; Mills *et al.*, 1993), which resulted in large numbers of marine alien species introductions, particularly in the Mediterranean Sea (Galil *et al.*, 2014). The extensions of these shipping canals (Galil, Boero, Fraschetti, *et al.*, 2015; Muirhead *et al.*, 2015), as well as the opening of new transport routes such as the Northern Sea routes through the Arctic Ocean due to climate change or the intensification of existing routes, have led to more introductions of marine alien species (Ascensão *et al.*, 2018; Miller & Ruiz, 2014). Sudden declines in newly recorded marine alien species towards the end of the reported time series have been frequently noted (Gollasch, 2006; Wolff, 2005), which are associated with lags in detection and reporting (Wolff, 2005).

Status

One of the few global studies of marine alien species revealed hotspots in coastal areas of the North-East Atlantic, Northern European Seas, the Mediterranean Sea, Hawaiian Islands, and New Zealand (Bailey *et al.*, 2020; **Box 2.10** for more details). Many of the reported established alien species belong to arthropods, fishes, molluscs, and algae (Bailey *et al.*, 2020; Gollasch, 2006). The recently launched database WRiMS (M. J. Costello *et al.*, 2021) revealed similar hotspots, although a direct comparison is difficult due to varying spatial resolutions. That said, many regions that appear to have low numbers of reported alien species (i.e., not "hotspots"), may in fact reflect more on the history and intensity of investigation rather than the intensity of invasion. Until 2019, the Galapagos Islands were reported to be invaded by only five marine species, but a re-investigation revealed a minimum of 53 marine alien species present in that Archipelago (Carlton *et al.*, 2019). Chile is reported to have low numbers of marine alien species, with various hypotheses offered to explain the low alien species richness (Neill *et al.*, 2020), one being low research intensity. Comparing studies of similar sampling areas such as marine bays or port regions revealed alien species numbers of similar ranges with most species found in San Francisco Bay, United States (234 species) (Cohen & Carlton, 1998) followed by the Chesapeake Bay, United States (116 species) (Ruiz *et al.*, 1997), Port Philip Bay, Australia (99 species) (Hewitt *et al.*, 2004), Pearl Harbor, Hawaii (69 species) (Coles *et al.*, 1999) and Coos Bay, Oregon, United States (60) (Ruiz *et al.*, 1997). Most of these numbers are, however, based on data that are more than 20 years old and higher alien species numbers can be expected now. For example, J. T. Carlton & Eldredge (2009) updated the Pearl Harbor number from 69 to more than 175 (many species were older invasions or of other taxonomic groups not noted in Coles *et al.* (1999), and thus not post-1999 invasions).

On the whole Hawaiian Archipelago, 333 marine alien species have been reported (Carlton & Eldredge, 2009, 2015). Among European Seas, by far the largest numbers of marine alien species have been recorded for the Mediterranean Sea (Galil *et al.*, 2021b; Katsanevakis *et al.*, 2020), followed by the North Sea and the Atlantic coast (Gollasch, 2006). Shipping (ballast water and hull fouling) and aquaculture have been consistently reported to represent the most important pathways for the introduction of marine alien species (Bailey *et al.*, 2020; Carlton & Eldredge, 2009; Coles *et al.*, 1999; Floerl & Inglis, 2005; Galil *et al.*, 2014; Gollasch, 2006; Hewitt *et al.*, 2004; Ruiz, Fofonoff, *et al.*, 2000; Schwindt *et al.*, 2020; Ulman *et al.*, 2019; S. L. Williams & Smith, 2007; **Box 2.10**). Often, large numbers of marine alien species are found at sites of intense human activity such as commercial ports (Ruiz *et al.*, 1997), marinas (Ulman *et al.*, 2019), or disturbed habitats (Coles *et al.*, 1999; S. L. Williams & Smith, 2007). Other vectors of introduction are fishing bait or ornamental purposes (Coles *et al.*, 1999; Gollasch, 2006). Patterns of distribution and trends were very similar across a wide range of taxonomic groups such as macroalgae, arthropods, cnidarian, polychaeta, molluscs, and fishes (Gollasch, 2006; Ruiz, Fofonoff, *et al.*, 2000; Seebens *et al.*, 2016; S. L. Williams & Smith, 2007). Microorganisms were frequently introduced (Cohen & Carlton, 1998; Ruiz, Rawlings, *et al.*, 2000); however, studies about the introduction of marine microorganisms and many other small size taxa are largely lacking.

Data and knowledge gaps

Among marine ecosystems, shelf ecosystems are much better investigated compared to the open ocean or the deep sea. Still, information about marine alien species remains one of the major gaps in the field of invasion ecology. Some high research interest regions such as North American

Box 2 10 Marine ecoregions: A global assessment of trends and status of alien and invasive alien species.

An extensive dataset of first detection records of marine alien species from 1965–2015 across 49 marine ecoregions is provided by Bailey *et al.* (2020). This dataset includes three major components of alien species records including the year of first collection, the invasion status, and potential pathways of introduction. Data were analyzed at both regional and global scales to examine the patterns of first record rate, species numbers, and transport pathways.

The assembled dataset included 2,209 records of marine alien species (1,442 unique species belonged to 17 phyla) where ten ecoregions had zero confirmed records during the period of study. On a global scale, about 75 per cent of marine alien species were reported as established and about 20 per cent had unknown invasion status, while the remaining records belonged to species with failed establishments (5.4 per cent) or extinct (0.5 per cent) populations. Most of the marine alien species were likely introduced as stowaways in ships' ballast water or biofouling. Escape of species from aquaculture or mariculture followed a similar pattern, while the corridor pathway and escape of pet or aquarium species increased beginning in the late 1990s. Nearly one-third of marine alien species' records were associated with a single pathway (32.7 per cent), while most were associated with at least two (52.6 per cent), or three (14.1 per cent) pathways. However, the patterns of alien species numbers varied across regions as a result of differences in pathway strength, environmental conditions, habitat size, survey effort, and taxonomic effort. The cumulative number of records from 1965-2015 ranged from zero to more than 500 per ecosystem, with various levels of succession of the population establishment across those regions. Ship fouling, transport stowaway, and ballast water were the dominant pathways in most regions, and were responsible for at least 40 per cent of introduction events. Other pathways became important for individual regions such as the corridor pathway (Suez Canal) in the Mediterranean Sea and escape of aquaculture/mariculture species in the East China Sea, South China Sea, and Yellow Sea (Bailey *et al.*, 2020). Although their dataset represents an extensive global collection of marine alien species records, it only covers about 73 per cent of the world's coastal large marine ecosystems, and data coverage was low in Africa, Meso- and South America, and Asia. As discussed in Bailey *et al.* (2020), marine alien species have undoubtedly occurred and reported in these areas, but due to cost of marine alien species surveys, limited resources, and lack of expertise across many taxa and regions, data of sufficient quality were likely not available for their study.

coastlines and European Seas, including the Mediterranean Sea, are comparatively well investigated, but data is far from complete and regular monitoring does not occur (Tsiamis *et al.*, 2021). Information for most other coastal areas is largely lacking. The most comprehensive available study on the global distribution of marine alien species shows large areas where information or expertise are lacking such as regions in Meso- and South America, Africa, and Asia (Bailey *et al.*, 2020). Even where information is available, lists are highly incomplete for many coastal areas. Based on expert knowledge, true numbers of marine established alien species might be up to ten times higher in some regions than reported in **Figure 2.5**.

2.5.4.2 Surface open ocean

Trends

Established alien species numbers are increasing in the open ocean from the tropics to polar regions due to warming oceans and human activity (M. J. Costello *et al.*, 2021). Many marine alien species tolerate a broader thermal range than native species and are able to show rapid physiological adaptation; both characteristics give alien species more habitat opportunities than natives (Canning-Clode *et al.*, 2011; H. Li *et al.*, 2020). For example, "Caribbean Creep" refers to a number of marine invertebrates (e.g., *Petrolisthes armatus* (green porcelain crab)) from the Caribbean that have expanded their distribution ranges poleward and invaded the southern and mid-Atlantic United States coasts (Canning-Clode *et al.*, 2011). Similarly, "African Creep" refers to the number of marine species moving poleward into the Mediterranean from lower latitudes (Canning-Clode & Carlton, 2017). In 1750, wooden sailing vessels could have carried 120 marine fouling and boring fauna and flora (Carlton, 1999b), while in the twentieth century, over 10,000 different marine species were estimated to be transported daily among different global geographic regions *via* ballast tanks (Carlton, 1999b) prior to the beginnings of detailed formulations for ballast water management. In this century, a vast global effort is underway to implement universal ballast water management strategies to prevent the transport and introduction of invasive alien species (**Chapter 5**, **section 5.5.1**).

The global rate of marine alien species records was relatively stable during 1965–1995 but increased significantly after 1995 and peaked at about 66 primary detections per year during 2005–2010, and then again decreased (Bailey *et al.*, 2020). Arthropods, molluscs, and fishes, by far the most thoroughly studied groups, were also not surprisingly the most frequently reported aquatic alien species during this time period and were most likely introduced as stowaways in ships' ballast water or biofouling. However, direct vector-related evidence was often absent. Arctic ship-based summertime transportation and tourism also increased over the past two decades, co-occurring with sea ice reductions (IPCC, 2019). This increase might bring implications for global trade and traditional shipping corridors economies, alerting the Arctic marine ecosystems and biodiversity, such as from invasive alien species and local pollution (IPCC, 2019). The relatively recent phenomenon of floating plastic debris in the open ocean facilitates the transport of coastal and oceanic species that might normally not survive the open ocean and may result in new and more frequent introductions of alien species across the oceans (**Chapter 3**, **section 3.3.3.3**; Haram *et al.*, 2021).

Environmental and anthropogenic changes have triggered reorganizations of reef ecology, zonation physiology, and dominance (Miranda *et al.*, 2020). One example is the plastic pollution in the ocean such as polystyrene foam which can be a dispersal vehicle for the invasive coral *Tubastraea* spp. (sun corals) (Faria & Kitahara, 2020). For example, in Brazilian reefs *Mussismilia harttii* (scleractinian coral) is threatened by the dominance of invasive sun corals (Faria & Kitahara, 2020; Miranda *et al.*, 2020). Sun corals lack natural predators and can reproduce rapidly with extensive defensive mechanisms which makes them a successful invasive alien species over large areas along the Brazilian coasts (Faria & Kitahara, 2020; Miranda *et al.*, 2020).

Status

There are more than 800 established alien species reported in the European seas only, some of which are invasive and impacting marine ecosystem services and biodiversity (Tsiamis *et al.*, 2018, 2020). Analyses revealed that a large number of alien species were not reported in initial assessments, or were proven to be historical misreporting (Tsiamis *et al.*, 2020). Thus, the Marine Strategy Framework Directive Descriptor 2 was implemented to provide an improved basis for reporting new alien species and to help the establishment of monitoring systems of targeted alien species (Tsiamis *et al.*, 2020). Major intentional introductions for fisheries also occurred with deep-sea species, such as *Paralithodes camtschaticus* (red king crab), native to the north Pacific coast and released in the Barents Sea during the 1960s (ICES, 2005). The species was later captured in the Ionian Sea in the Mediterranean (Faccia *et al.*, 2009), possibly transported by ballast water, though Faccia *et al.* (2009) raised doubts about whether a larva/post-larve presumably arrived in ballast water could withstand summer temperatures for so long – the specimen collected weighed about 4 kg and the estimated age was 10 years. Among tropical marine regions, Hawaii was found to be heavily affected by alien species either due to its location, governance (**Glossary**), or research effort undertaken to understand biological invasions in this region (Alidoost Salimi *et al.*, 2021). An alternative explanation might be also due to lower native biodiversity associated with Hawaiian ecosystems providing more vacant niches being available to the alien species.

The recently launched WRiMS (marinespecies.org/introduced) is an expert-edited world list of introduced marine species and provides information of alien and invasive alien marine organisms (M. J. Costello *et al.*, 2021). An alien marine metazoan species checklist for the Mediterranean Sea lists 573 alien species (Galil *et al.*, 2014). Most of those alien species are thermophilic, originally from the Indo-Pacific or Indian Oceans that invaded the Mediterranean through the Suez Canal (Galil *et al.*, 2014). Additionally, the Information System on Aquatic Non-Indigenous and cryptogenic Species (AquaNIS) database provides information on 859 aquatic alien and cryptogenic species in the North Atlantic region (AquaNIS, 2015).

Data and knowledge gaps

The open sea represents one of the least investigated units of analysis with respect to biological invasions. The size and cost of sampling the open sea presents a particular challenge. Another challenge is how "alien" is defined in the open sea because it is usually defined for much smaller geographic units such as countries – a challenging concept to transfer to the open ocean. Some databases, such as WRiMS (M. J. Costello *et al.*, 2021), also cover the open ocean, but the vast majority of records have likely been sampled along the coasts. However, WRiMS records provide the opportunity to map the actual locations of marine alien species using records from the Ocean Biodiversity Information System (OBIS) or GBIF. Nonetheless, a comprehensive assessment of the trends and status of alien and invasive alien species in open oceans is still missing and difficult to conduct currently due to the lack of records.

There are other global databases of species occurrences such as AlgaeBase (Guiry & Guiry, 2021) or FishBase (Froese & Pauly, 2015), but the information about the status of invasion is incomplete or totally lacking. There are also distributed occurrence records for marine alien species in the GRIIS dataset (Pagad *et al.*, 2022) and other national checklists, but these usually reflect coastal areas rather than occurrences in the open ocean. This lack of information on open ocean alien species occurrences represents one of the largest knowledge gaps across all units of analysis.

2.5.4.3 Deep sea

Trends

As biota occurring at deep ocean depths have been rarely surveyed (Saeedi, Costello, *et al.*, 2019; Saeedi *et al.*, 2020; Saeedi & Brandt, 2020), there are too few records over too short a time period to infer trends. The deep-sea populations of alien species may follow a "boom-and bust" pattern of abundance (Strayer *et al.*, 2017), such as documented between 1995-2002 for *Philine auriformis* (New Zealand sea slug) in southern California, United States (Cadien & Ranasinghe, 2003), settle for long-term low-abundance stability, or, following a time lag or environmental triggering event, result in greatly increased abundance. As depth increases, less measurements are available for biological variables (M. J. Costello *et al.*, 2018; Saeedi, Bernardino, *et al.*, 2019), making estimations of rates of biological invasion challenging in the deep ocean.

Status

Records of biological invasions into depths greater than 200 meters are rare. The intentional introduction of the economically important North Pacific *Paralithodes camtschaticus* (red king crab) in the 1960s into the Barents Sea demonstrated that the deep ocean is not immune to invasions (Dvoretsky & Dvoretsky, 2018; Jørgensen & Nilssen, 2011). Immature individuals remain on the shallow shelf (20–50 m), adult specimens mostly inhabit deep soft-bottom areas (100–400 m), migrating into shallow waters (less than 50 m) for moulting and mating (Sundet & Hjelset, 2011). Specimens of *Pterois* spp. (lionfishes) that invaded the Western North Atlantic/Caribbean region were reported from Bermuda, Curaçao, and Honduras at depths between 250 and 300m (Andradi-Brown, 2019). *Philine auriformis* (New Zealand sea slug) was introduced to the West Coast of North America (southern California, Untied States of America, to British Columbia, Canada) and occurs from the intertidal to more than 300 m (Cadien & Ranasinghe, 2003). In the south-east Mediterranean Sea, four carnivorous Red Sea species, *Champsodon nudivittis* (crocodile toothfish), *Etrumeus golanii* (Golani's round herring), *Trypauchen vagina* (burrowing goby), and *Charybdis longicollis* (lesser swimming crab) were recently recorded at depths over 200 m (Galil *et al.*, 2019; Innocenti *et al.*, 2017). One possible pathway of deep-sea species translocations may be deep submergence vehicles whose use has increased since the 1960s (Voight *et al.*, 2012). It seems realistic to suggest that understanding the scale of deep-sea invasions by alien species remains one of the most important overlooked aspects of marine invasion science.

The deep sea is now also warming, as has been observed in shallow waters, and the temperature of water below 2000 m has increased since 1992, especially in the Southern Ocean (IPCC, 2019). For example, deep Mediterranean waters have warmed by 0.12 °C since the mid-twentieth century and the deep oceans now store 16–89 per cent more heat than before (McClain *et al.*, 2012). Temperature changes and the redistribution of total energy will ultimately impact deep-sea faunal distributions and invasion rates. For example, some deep-sea fish families of Actinopterygii were identified with depths over 1000 m and were proposed as invasive alien species where most of their constituent species live in shallower than 1000 m (Priede & Froese, 2013). Also, the invasion of Erythrean species of the

Levantine basin into the lower continental shelf and upper slope suggests biological invasions in the deep sea warrant more attention (Galil *et al.*, 2019). The west Antarctic Peninsula shelf is rapidly warming and is expected to soon be invaded by lithodid crabs from the Ross Sea waters that have crossed the Antarctic shelf (C. R. Smith *et al.*, 2012).

Data and knowledge gaps

Estimating the gaps in alien species distributions of the deep-sea fauna is challenging because the deep sea is the most unexplored place on Earth and there is much yet to be learned. However, alien species pose a threat to the unique, diverse, and fragile mesophotic "animal forests". Large data and knowledge gaps therefore remain for trends and status of invasive alien species in the deep sea as well as a lack of information the actual data gaps.

2.5.5 Trends and status of alien and invasive alien species in anthropized areas

2.5.5.1 Urban/semi-urban

Urban habitats include constructed, industrial, and other artificial land, human settlements, buildings, industrial developments, transport networks and waste dump sites, but also a diversity of semi-natural and constructed green spaces. Cities contain high densities of people and are hubs of human-mediated movement of commodities. Transport linkages (e.g., airports and harbours) facilitate the introduction and dissemination of alien species through introduction pathways such as trade, tourism, and horticulture (**Chapter 3**, **section 3.2.3**; Dehnen-Schmutz *et al.*, 2007). The intensive study of alien plants in urban areas began in a few cities around the world in the 1980s (Esler, 1987; Kowarik, 1990; Stalter *et al.*, 1992), largely out of natural history interest. Large-scale comparisons of alien plant taxa among cities grew out of a more macroecological approach in Europe in the 1990s (Kowarik, 1995a; Pyšek, 1998), which has since given way to more recent global assessments of patterns of alien species in cities (Aronson *et al.*, 2014; Gaertner *et al.*, 2017).

Trends

Evidence suggests that the rate and extent of biological invasions are increasing globally (Seebens, Blackburn, *et al.*, 2017) and cities often play important roles as hubs for the spread of alien species (Chytrý *et al.*, 2012; McLean *et al.*, 2017). Studies on long-term dynamics of urban floras revealed a steep increase in established alien species numbers along with accelerating urbanization during the last century (Chocholoušková & Pyšek, 2003; S. Knapp *et al.*, 2010; Tretyakova *et al.*, 2018), with alien species occupying a median of 28 per cent (ranging from 25-50 per cent) of their respective urban floras (Aronson *et al.*, 2014; Esler, 1987; Ricotta *et al.*, 2009, 2012; G.-L. Zhu *et al.*, 2019). Several studies from around the world show that more urbanized areas tend to harbour a higher relative abundance and diversity of alien species than rural and peri-urban areas (Aronson *et al.*, 2015; Blair & Johnson, 2008; Cadotte *et al.*, 2017; X. Chen *et al.*, 2014; Lowry *et al.*, 2020), and as urbanization expands, the numbers of alien taxa in urban areas will consequently increase as well.

Projected trends in plant invasions in Europe under different scenarios of future land-use change showed the second highest level for urban areas (Chytrý *et al.*, 2012). Most alien species in cities and urban areas are intentionally introduced ornamental plants that escaped from cultivation (Čeplová *et al.*, 2017; Dehnen-Schmutz *et al.*, 2007; McLean *et al.*, 2017; Padayachee *et al.*, 2017). Studies in the Czech Republic, for example, reveal that 47 per cent of alien species now found in cities and beyond were introduced intentionally, mostly as ornamentals (Pyšek *et al.*, 2002), and work from South Africa showed that twice as many of the most abundant alien species in urban areas were originally introduced for ornamental purposes compared to non-ornamental alien species (McLean *et al.*, 2017). Much like agriculture, plantings of alien plants in urban settings provide suitable habitats for the establishment of alien insects; consequently, urban settings and especially street trees tend to be hotspots for insect invasions (Branco *et al.*, 2019; Dale & Frank, 2017; Paap *et al.*, 2017).

It is likely that a warmer climate together with urban sprawl will increase the invasion risk for cities, especially as species from different climatic regions are transported elsewhere, and especially from warm regions to temperate ones (e.g., Géron *et al.*, 2021; Lososová *et al.*, 2018). For Europe, Lososová et al (2018) suggest that alien species from regions with warm climates, such as those currently limited to southern Europe, are likely to increase their rate of spread and colonize the cities of Central and Western Europe. Alien insects appear to be especially benefiting from increased urban temperatures, for example, alien mosquitos in montane cities in South America (Pedrosa *et al.*, 2020) and alien scale insects in the United States (Meineke *et al.*, 2013).

Status

The most comprehensive global data set on urban floras and bird faunas, based on 110 and 54 cities on all continents, respectively, revealed that the numbers of alien species differ broadly among cities with a median of 3.5 alien bird (range: 0–23) and 213 plant species (range: 38–1058), of the total species richness 112.5 (range: 24–368) for birds and 766 (range: 269–2528) for plants. Among plants, Poa annua (annual meadowgrass), Capsella bursa-pastoris (shepherd's purse), Stellaria media (common chickweed), Plantago lanceolata (ribwort plantain), and Phragmites australis (common reed) have established in the greatest numbers of cities, while among birds such species are Columba livia (pigeons), Passer domesticus (house sparrow), Sturnus vulgaris (common starling), and Hirundo rustica (barn swallow) (Aronson *et al.*, 2014). Further, it appears that intensive land-use change, and biotic interchange have increased the similarity of urban plant assemblages globally. Cities in disparate regions of the globe thus retain regionally distinct native and alien plant assemblages (Palma *et al.*, 2017), while invasive alien species are associated with lower beta diversity among cities (La Sorte *et al.*, 2014).

The numbers of established alien species of plants, insects, herptiles, birds, and mammals, introduced to Europe after 1500 and occurring in habitats defined according to the European Nature Information System were analysed for 115 regional data sets (Pyšek, Bacher, *et al.*, 2010). Cities in Europe on average harbour 70 per cent of established alien plants (ranging from 41–100 per cent in individual regions), 54 per cent (11–76 per cent) of alien insects, 38 per cent (0–100 per cent) of alien herptiles, 14 per cent (0–33 per cent) of alien birds, and 26 per cent (0–100 per cent) of alien mammals. The numbers of established alien plant and insect species found in human-made, urban, or cultivated habitats were the highest of all habitats, if controlled for habitat area in the region (Pyšek, Bacher, *et al.*, 2010). The patterns of urban alien diversity have not been summarized beyond Central and Western Europe, but studies from elsewhere, for example, China, Russia, and Canada, also confirm that urban areas tend to contain very high numbers of alien species (Cadotte, 2021; Tretyakova *et al.*, 2018; Z.-X. Zhu *et al.*, 2019).

Data and knowledge gaps

Although urban ecosystems are hotspots for biological invasions, the field of invasion science has given scant attention to invasion dynamics in towns and cities (Gaertner *et al.*, 2017) with the exception of Europe where this topic has been subject of research for decades (e.g., Kowarik, 1995b; Pyšek, 1998; Sukopp, 2002). Many facets of biological invasions require elaboration in an urban context (Cilliers *et al.*, 2008; Padayachee *et al.*, 2017). The role of cities as launching sites for alien species introduction and spread into natural areas and as recipients of a range of socioecological impacts highlights the need for research to address key limitations that hinder the understanding of invasion dynamics in urban settings. There have been very few urban-rural gradient studies in developing countries (Pauchard *et al.*, 2006), or in tropical environments in general (Cusack & McCleery, 2014). So far, the relationship between levels of urbanization and abundance of alien invasive plants in tropical developing countries appears to resemble that of temperate developed countries (Lowry *et al.*, 2020). Limitations include the dearth of metrics for defining urban–wildland/rural gradients and a shortage of insights on many aspects of urban invasions in less affluent regions (Gaertner *et al.*, 2017). Thus, data on alien taxonomic groups other than plants within cities and ecoregions surrounding each city is needed.

2.5.5.2 Cultivated areas (including cropping, intensive livestock farming, etc.)

Many introductions and secondary spread of alien species occur in cultivated areas. Alien plant species that occur as weeds in agricultural areas can be introduced as contaminants of seeds, or spread by machinery and grazing animals, water channels, etc. In addition, the use of plant protection products may promote the development of herbicide resistant alien weeds, as in the case of Amaranthus, Solanum, etc. In addition, agricultural areas are often first sites of new introduction of novel crops, genetically modified organisms, biofuel crops, and novel genotypes of cultigens. In some parts of the world, ornamental plants are also intensively cultivated in agricultural areas (e.g., Booth *et al.*, 2003). Cultivated plants also suffer from introduced pathogens (e.g., fungal, viral, bacterial).

Various pathways are known to facilitate the accidental introduction of insects, pathogens, and other pests (e.g., nematodes) into cultivated areas around the world. Many groups of insects colonize stored grains and international trade in grain has facilitated the global spread of these insects such that several important species are established in virtually every world region (Morimoto *et al.*, 2019). Other important pathways by which insect pests have globally spread include international trade in fruits and vegetables and global transport of live plants, including soil and planting substrates (Kiritani & Yamamura, 2003; Liebhold *et al.*, 2012). Prior to 1910, there was little recognition of the dangers that such international trade posed for introduction of agricultural pests, but in the early 1900s many countries began to implement regulations aimed at limiting the accidental spread of plant pests with plants and plant parts. A variety of phytosanitary measures have been developed to limit pest movement in international trade, though some pathways remain more difficult to control and many species continue to be unintentionally introduced (E. Allen *et al.*, 2017; Hulme, 2014; **Chapter 5**, **section 5.2.2**).

Trends

Reports on occurrences of alien species on cultivated land are usually restricted to plant pathogens, while more general comprehensive analyses of trends of alien species on cultivated areas are largely lacking. For alien species considered as plant pathogens, which mostly consist of arthropods, fungi and oocmycetes, the number of species has increased continuously since 1800 with a rise also in the rate of annual records until the present (Aukema *et al.*, 2010; Kiritani & Morimoto, 2004; Nealis *et al.*, 2016; R. M. Smith *et al.*, 2018; F.-H. Wan & Yang, 2016). This is very likely a result of increased trade activity, particularly of plant materials, both in terms of increased volumes and increased geographic distances between donor and recipient regions. While the number of studies is geographically restricted to a few well-sampled regions, global analyses are missing; however, it is likely that alien species numbers have been increasing as observed in other world regions.

Status

Agricultural areas in Eastern Europe are the most invaded by alien plants of all European regions (Chytrý *et al.*, 2009). On arable land there were on average 7.3±9.8 per cent of plant species introduced after 1500 in Catalonia (n=506), 5.6±5.2 per cent in the Czech Republic (n=1441) and 14.3±25.6 per cent in the United Kingdom (n=989); these values represent per centages of all plants recorded in vegetation plots 15–200 m^2 in size (Chytrý *et al.*, 2008). For plants introduced from the beginning of Neolithic agriculture until 1500 (Pyšek & Jarošík, 2005), 55.5±13.5 per cent and 16.2±16.0 were reported for the Czech Republic and the United Kingdom, respectively (Chytrý *et al.*, 2008).

Data from cultivated habitats in Europe comparing alien species of plants, insects, herptiles, birds and mammals introduced after 1500 showed that as a per cent of the total alien species in a region, cultivated habitats on average harbour 34 per cent of plants (based on 115 regional datasets: median, with range 5–95 per cent), 46 per cent (26–66 per cent) of insects, 63 per cent (0–100 per cent) of herptiles, 65 per cent (51–85 per cent) of birds, and 30 per cent (0–100 per cent) of mammals (Pyšek, Bacher, *et al.*, 2010). By this measure, cultivated habitats are among those with the highest levels of established alien species (Pyšek, Bacher, *et al.*, 2010).

The domestication of plants and their widespread planting in agriculture has created unique resources that facilitate the establishment of new insect species (Liebhold *et al.*, 2018). Across most continents, the historical expansion of plantings for agriculture and forestry has been followed by the invasion of insects that utilize these crop species as hosts (e.g., Hurley *et al.*, 2016; Margaritopoulos *et al.*, 2009).

Data and knowledge gaps

Information on biological invasions of insects and plants in cultivated areas has been systematically collected in Europe and North America, likely because they act as pests and weeds and negatively impact agricultural production. However, information from other parts of the world is scarce.

2.5.5.3 Aquaculture areas

Inland, coastal, and marine farming is largely based on introduced species and a large share of the industry occurs in South-East Asia and South America. In addition to being an important pathway of introduction for alien species, aquaculture facilities can also contain many pathogens, parasites, and fouling species unintentionally introduced as contaminants with the farmed species and the materials used for their production (e.g., K. E. Costello *et al.*, 2021; Peeler *et al.*, 2011). Molluscs can carry many non-target species with them: for example, several introduced marine algal alien species worldwide were transported in association with mariculture, mainly of molluscs (Mckindsey *et al.*, 2007). In Europe, the production of native oyster *Ostrea edulis* (European oyster) has been greatly impacted by the parasite protozoan *Bonamia ostreae*, one of the diseases notifiable to the World Organisation for Animal Health (WOAH, founded as OIE; Carnegie & Cochennec-Laureau, 2004), and also by the parasitic copepod *Myicola ostreae*, both introduced together with *Magallana gigas* (Pacific oyster) (K. E. Costello *et al.*, 2021). Two bivalves (*Magallana gigas*, and *Ruditapes philippinarum* (Japanese carpet shell)) were responsible for the majority of introductions of contaminants in Europe (60 species), mainly shell foulants or macroalgae used for packaging live oysters and clams (Savini *et al.*, 2010). The aquaculture of *Magallana gigas* is likely responsible of the introduction of *Styela clava* (Asian tunicate) in New Zealand, which poses a threat to the shellfish aquaculture industry (Forrest *et al.*, 2011). Many alien species introduced for aquaculture have escaped from confined systems, established, and become invasive (Ju *et al.*, 2020): for example, the analysis of both marine and estuarine species in California showed that 106 of 126 (84 per cent) introductions were due to aquaculture and led to established populations of alien bivalves (K. E. Costello *et al.*, 2021).

Trends

Worldwide, the introduction of alien species in aquaculture is well-known, but the numbers have significantly increased since the 1950s with technological improvements (i.e., development of artificial propagation, (Shelton & Rothbard, 2006)). Other notable increases were reported in the 1960s and 1970s with the movement of *Tilapia* spp. (tilapia) and *Oreochromis* spp. (tilapia). In the 1990s Asian carp (e.g., *Ctenopharyngodon idella* (grass carp), *Hypophthalmichthys nobilis* (bighead carp), *Hypophthalmichthys molitrix* (silver carp)) was used to meet the growing demand of food to reduce the harvesting of wild species and to diversify the production (De Silva, 2012; De Silva *et al.*, 2006; Naylor *et al.*, 2001; Shelton & Rothbard, 2006). This increasing trend is consistent across the continents (FAO, 2020), particularly in Asia. China, for example, has experienced a notable increase of alien species farmed in aquaculture mostly in the 1990s, even though the introductions started in the 1920s

(Casal, 2006; Cook *et al.*, 2008; Y. Lin *et al.*, 2015; J. Liu & Li, 2010; Q. Wang *et al.*, 2015; Xiong *et al.*, 2015, 2017). A similar increase was reported for Europe beginning in the 1970s (Olenin *et al.*, 2008; Savini *et al.*, 2010; Turchini & De Silva, 2008), and in the Americas (Gozlan, 2008), especially in Latin America and the Caribbean since the 1970s-1980s with the introduction of salmonids, tilapia, Asian carps and shrimps (Shelton & Rothbard, 2006). In the United States, many native species are cultured for food, and tilapia and Asian carp introduction for food production began in the 1950-60s (Shelton & Rothbard, 2006). In Africa, aquaculture production increased since the 1980s (Shelton & Rothbard, 2006), relying mainly on introduced Asian carp and African tilapia moved within the African continent (Bartley & Marttin, 2004). In Africa, three waves of fish introductions (a total of 139 species, 40 per cent for aquaculture) occurred: before 1949, between 1950-1989, and after 1990 (Satia & Bartley, 1998). In Oceania, even though few alien species were introduced for aquaculture since 1900, this region began having an important position in aquaculture production during the 1970s (Gozlan, 2008), with alien species making up 38 per cent of the production on average (Cook *et al.*, 2008). Overall, aquaculture is mainly for food production. However, the market for ornamental and angling species is increasing, especially in Asia, Europe, and North America, thus increasing aquaculture-based introductions for this purpose (reviewed in Gozlan, 2008). Indeed, in the United States, more than half of the 91 fish species introduced through aquaculture are ornamental (J. E. Hill, 2008).

Fish, molluscs, and crustaceans are the most introduced taxonomic groups in aquaculture. Aquaculture is responsible for the majority of fish introductions globally (De Silva *et al.*, 2009; Teletchea, 2019), as confirmed by the positive correlation shown between aquaculture production and the number of fish species introduced to a region (Gozlan, 2008). Overall, the introductions of fish started before the other groups, with a first "wave" before 1900, followed by other waves in the early 1900, after 1950 and after 1960s-70s (Shelton & Rothbard, 2006): Casal (2006), extracting the data of FishBase, reported 3072 fish introductions involving 568 species, with aquaculture being the main reason of introduction (40 per cent), while in 2008, Gozlan (2008) mentioned 624 fish species introduced worldwide, 51 per cent of them for aquaculture. Freshwater fish, particularly *Cyprinus carpio* (common carp), tilapia (specifically *Oreochromis niloticus* (Nile tilapia) is the main farmed tilapia), *Salmo trutta* (brown trout), and *Oncorhynchus mykiss* (rainbow trout) are the most introduced for aquaculture production (De Silva, 2012; Teletchea, 2019). Only 15 marine fish have been introduced for aquaculture (Atalah & Sanchez-Jerez, 2020). In contrast, all molluscs introduced for aquaculture are marine (19 species reported in (De Silva, 2012; X. Guo, 2009), with *Magallana gigas* (Pacific oyster) being one of the most successfully introduced aquatic alien species throughout the world since the end of nineteenth century in United States, Canada, Europe, Australia, New Zealand, Mexico, Peru, Chile, Argentina, and South Africa (De Silva, 2012; X. Guo, 2009). The other alien mollusc species were mostly introduced in the 1960s and from the 1980s (X. Guo, 2009). In the last twenty years, the most widely introduced alien species were reported from the eastern Pacific, such as *Penaeus vannamei* (whiteleg shrimp) reported by Fernández de Alaiza García Madrigal *et al.* (2018); in 2013, its production of 4.3 million tons represented 64 per cent of the global farmed shrimp production. Finally, since the 1970s, many alien seaweeds have been unintentionally introduced through aquaculture, while very few species were intentionally introduced for production (FAO, 2020; Pickering *et al.*, 2007).

Status

Asia is considered the "backbone of global aquaculture production" (De Silva, 2012) with its contribution to over 90 per cent to the sector (De Silva *et al.*, 2009); aquaculture heavily relies on alien species (De Silva *et al.*, 2006, 2009; Ju *et al.*, 2020), particularly, in China, the leading global aquaculture producer (more than 60 per cent of the global production, Cao *et al.*, 2015; Q. Wang *et al.*, 2015). In China, alien species (a total of 179 species, Y. Lin *et al.*, 2015) are involved for over 25 per cent of the total production (Xiong *et al.*, 2017), compared to the 17 per cent of global production of alien species (Shelton & Rothbard, 2006). Asia also stands out for the widely cultured species of *Penaeus vannamei* (whiteleg shrimp), introduced in 1978 in Asia, with contributions from China, Thailand, Indonesia, and Vietnam to most of the world's shrimp production (Liao & Chien, 2011). In Europe, at least 703 alien species introduced to aquatic ecosystems for aquaculture and stocking activities have been reported: fish, crustaceans and molluscs are the most introduced taxonomic groups (Olenin *et al.*, 2008; Savini *et al.*, 2010; Teletchea, 2019; Turchini & De Silva, 2008). In Europe, alien species (mostly *Oncorhynchus mykiss* (rainbow trout), *Hypophthalmichthys molitri* (silver carp) and *Cyprinus carpio* (common carp)) contributed 67 per cent of freshwater aquaculture production, mainly in Western areas with a range of 88-98 per cent (Turchini & De Silva, 2008). The highest production of introduced marine fish is concentrated in the Magellanic province of southern Chile that is considered at risk of environmental impacts caused by escapees from the confined environment (Atalah & Sanchez-Jerez, 2020). Recent planning for diversification in aquaculture reports advised for a shift towards producing more native than alien species (Harvey *et al.*, 2017).

The worst impacts on aquaculture production have been caused by the oomycete *Aphanomyces astaci*, the causative agent of the crayfish plague. Vectored by North American crayfish introduced to Europe for aquaculture, this plague dramatically reduced native populations and the production of native European crayfish (De Silva *et al.*, 2009). Many pathogens can also be carried by alien finfish, especially cyprinids: at least 226 parasite species (34 of which causing important diseases worldwide) have been found in *Cyprinus carpio* (common carp), one of the most introduced alien species (Jeney & Jeney, 1995). In Europe, the seven most farmed cyprinids led to the introduction of 31parasites⁄disease agents (Savini *et al.*, 2010). Similarly, in South Africa many parasites have been introduced with fish and crayfish used for fisheries and aquaculture (Weyl *et al.*, 2020). Despite the high number of pathogens transferred by alien farmed fish, a large-scale mass mortality of farmed fish due to introduction of associated pathogens has not yet been recorded (De Silva *et al.*, 2009). Still, alien farmed shrimps can carry several diseases that lead to important outbreaks in the facilities and relevant economic losses, especially in Asia (Briggs *et al.*, 2004).

Data and knowledge gaps

The Food and Agriculture Organization (FAO) Database on Introductions of Aquatic Species (DIAS) (FAO, 2021) reports the introduction of alien species per country, providing also global maps of species introduced for aquaculture and a focus on some alien species, such as *Cyprinus carpio* (common carp) and *Oreochromis niloticus* (Nile tilapia). In general, there is considerable information available for Asia, the leading continent for aquaculture production, and for Europe and Latin America while for other regions information is often lacking. Recent reviews addressed fish, molluscs and shrimp situations. Studies on temporal trends are limited and mainly available for the three main taxonomic groups fish, molluscs, and crustaceans.

2.5.5.4 Coastal areas intensively used for multiple purposes by humans

Trends

Accumulation rates of established alien species in coastal marine waters frequently show a pattern of exponential accumulation through time, with the number of new reports increasing dramatically during the last 30 years with increased awareness and research effort (Bailey *et al.*, 2020; Leppäkoski *et al.*, 2002; Ruiz *et al.*, 2015). The earliest substantiated reports of established alien marine species date to at least the 1200s (Ojaveer *et al.*, 2018). The type of transported taxa has changed over time as shipping pathways have modernized. For example, historical use of solid ballast, such as rocks, sand, and dirt, was associated with the transportation of seeds and insects while the modern use of seawater ballast correlates with introductions of aquatic taxa ranging from microbes and protists to macroinvertebrates and fishes (Bailey, 2015). There are also now fewer intentional introductions of fishes and macroinvertebrates into the natural environment, likely because the potential negative impacts of such releases are now better understood (Bailey *et al.*, 2020).

While the rate of new alien species records has levelled off and even declined since 2010, possibly due to regulations for ships' ballast water and improved practices by the aquaculture industry (Bailey *et al.*, 2020; **Chapter 5**, **section 5.51**), expectations of continued global shipping growth suggest the risks of biological invasions could increase significantly by 2050 without management of shipping-mediated vectors (Sardain *et al.*, 2019) thus underscoring the importance of existing instruments to prevent introductions *via* ballast water and biofouling. The construction and successive enlargement of canals connecting previously unconnected waterbodies has been responsible for a growing number of established alien species in the Mediterranean (Galil *et al.*, 2017). Similarly, it has been projected that the recent expansion of the Panama Canal could triple the number of established alien species arriving in the Gulf of Mexico and the North American East Coast (Muirhead *et al.*, 2015). In regions such as the Arctic, the changing environmental conditions and the dramatic increase in shipping activity are likely to favour the transport and introduction of new alien species. This increase in alien species is likely to reconfigure the global dynamics of invasive alien species, potentially reshaping marine habitats and ecosystem functions, especially in coastal regions (Goldsmit *et al.*, 2020; Miller & Ruiz, 2014).

Status

There has been extensive research and surveillance of coastal marine alien species in Central and Western Europe, with more than 4,350 detection records for at least 1,370 introduced species of alien or unknown (cryptogenic) origin (AquaNIS, 2015). More than 450 marine alien species have been recorded off the Israeli Mediterranean coast – which serves as a gateway for introductions from the western Indian Ocean and Red Sea, through the Suez Canal, to the Mediterranean Sea (Galil *et al.*, 2021a).

Coastal areas are generally prone to biological invasions. In a global study of established alien species richness of a number of taxonomic groups, Dawson *et al.* (2017) found that hotspots are, other than islands, predominantly coastal mainland regions.

In the Americas, at least 450 alien species are reported from continental North America (Ruiz *et al.*, 2015), and approximately 300 other species from Hawaii (Carlton & Eldredge, 2009). Reported numbers are lower in South America, with 129, 138, and 53 species reported from the south-west Atlantic, Brazil, and the Galápagos Islands, respectively (Carlton *et al.*, 2019; Schwindt *et al.*, 2020; Teixeira & Creed, 2020). Despite the low number of reported alien species, the coastal environments of the south-west Atlantic were affected by one of the largest continental-scale bioinvasion events ever recorded, and which has reshaped vast coastal-marine ecosystems, modifying their coastal geomorphology, biodiversity, primary and secondary productivity in the Americas and Asia (Bortolus *et al.*, 2015, 2019; Qiu, 2013). Researchers have shown that what are now extensive *Sporobolus alterniflorus* (smooth cordgrass) marshes in this region, were probably bare mudflats centuries ago, and that the *Sporobolus alterniflorus* introduction might have led to vast unrecorded shifts in bird, fish, and invertebrate biodiversity, and immense shifts in algal vs. detritus production, with the concomitant trophic cascades that these changes imply (Bortolus *et al.*, 2015, 2019). Reports of mudflat conversion by *Sporobolus alterniflorus* with distinct ecological consequences have also been reported from China (B. Li *et al.*, 2009). Similarly, the coastal systems of North America have been transformed by an introduced genotype of the macrophyte *Phragmites australis* (common reed) causing whole ecosystem and habitat transformations (Bowen *et al.*, 2017; Chambers *et al.*, 1999; Cronin *et al.*, 2015; Dibble & Meyerson, 2014).

In the Asia-Pacific region, at least 650 marine alien and cryptogenic species are reported from New Zealand (Seaward & Inglis, 2018), with another 343 introduced and cryptogenic species reported from Australia (Sliwa *et al.*, 2008), and 213 alien species reported from China (Xiong *et al.*, 2017). At least 95 alien and 39 cryptogenic species are reported from South Africa (T. B. Robinson *et al.*, 2016), with most of the African continent being understudied.

From1965-2015, at least 1,400 unique alien species have been reported as being introduced in coastal ecosystems – approximately one new species detected every 8 days for the last fifty years (Bailey *et al.*, 2020).

Data and knowledge gaps

Records of alien species in coastal environments are more reliable in recent decades as the awareness of alien species introductions and their potential negative impacts began to increase. However, data are still limited for many taxonomic groups and regions of the world (especially Africa, Meso- and South America and Asia) (Bailey *et al.*, 2020). Aquatic alien species are frequently under-reported due to limited research intensity and insufficient taxonomic expertise (especially for smaller-bodied organisms) (Carlton & Fowler, 2018; Ojaveer *et al.*, 2017). Reliable records of alien species introductions exist mainly for plants and animals, with fungi, protists, and microbes generally being understudied.

An accurate number of alien species introduced across global coastal waters is difficult to estimate since organisms were being transported around the world by ships for centuries before inventories of species in the marine environment, resulting in an inability to determine the true origin of a large proportion of species within coastal communities (Bortolus *et al.*, 2015; Carlton, 1996; Hewitt *et al.*, 2004; Schwindt *et al.*, 2020). There can also be long time lags after the initial introduction and establishment of a new population until its discovery (C. J. Costello & Solow, 2003; C. M. Taylor & Hastings, 2005), unless regular and targeted monitoring is taking place (Hayes *et al.*, 2019). In many regions of the world, regular surveillance is hampered by inadequate resources and limited access to taxonomic expertise (Ojaveer *et al.*, 2014). The number of alien introductions is therefore certainly much higher than published literature suggests.

The study of invasive alien vascular plant species introduced in the marine-coastal environments of South America is currently one of the largest gaps to cope with. Besides a few classic examples including genera such as *Tamarix* (tamarisk), *Carpobrotus, Ammophila, Sporobolus*, or *Salsola* (Schwindt *et al.*, 2018), there is little research effort in this area and no updated review or synthesis revising the list of plant invasive alien species for this region. Large regions like South America have invested little effort (e.g., relative to Europe or North America) to recording and monitoring the introduction of alien species. This lack of data has often been misunderstood as an actual lack of invasive alien species. This knowledge gap seriously hampers the ability to recognize pre-existing native ecosystems (i.e., Ecological Mirage Hypothesis; Bortolus *et al.*, 2015; Bortolus & Schwindt, 2007). On the other hand, there is currently an increase in the number of researchers investigating invasive alien species in this region (Schwindt & Bortolus, 2017), which will likely increase the number of reports of introduced species for the region. Nevertheless, this increase is not necessarily, or strictly, due to new introductions, but could also include introductions long overlooked and ignored. For instance, in 2017 scientists found that what was until then considered a native alga, *Melanothamnus harveyi* (Harvey's siphon weed), was in fact the earliest record of an alien coastal marine species for the region, being first reported in 1872 under the name of *Polysiphonia argentinica* (Schwindt *et al.*, 2020). Similarly, *Sporobolus alterniflorus* (smooth cordgrass) was recognized as alien to the southern Atlantic coastal environments by 2015, nearly two centuries after its introduction (Bortolus *et al.*, 2015).

Finally, the lack of research on emerging or understudied transportation pathways, such as the aquarium and bait trades, internet commerce and anthropogenic marine litter (e.g., M. L. Campbell *et al.*, 2017; J. T. Carlton *et al.*, 2017; Fowler *et al.*, 2016; Lenda *et al.*, 2014), likely results in gaps of knowledge. This knowledge gap refers to the relative importance of different introduction mechanisms and the corresponding management priorities for reduction of future introductions of aquatic alien species.

Box 2 11 Good Quality of Life: A global assessment of trends and status of invasive alien species.

Invasive alien species are a significant and growing threat worldwide to the good quality of life (i.e., the achievement of a fulfilled human life, see IPBES glossary[6] for a complete definition) for many communities (Costanza *et al.*, 2006). A literature review conducted by the authors of **Chapter 4** identified about 1050 invasive alien species that impact good quality of life (**Chapter 4**, **Figure 4.2**). In most cases (841 cases), the reported impacts negatively affected good quality of life, while in 212 cases, benefits of invasive alien species were reported. However, it is critical to note that a benefit from an invasive alien species in one sector does not mitigate the harm caused elsewhere, and that the same invasive alien species may both cause harm and produce a benefit. Integrating this invasive alien species list and the distributional data provided in this chapter (**section 2.1.4** for data details) reveals that the United States, Australia, New Zealand, multiple European countries, China, Japan, Canada, Mexico, and South Africa were the countries with highest numbers of invasive alien species with impacts (negative or positive) on the good quality of life **(Figure 2.36)**. This pattern largely reflects the distribution of all identified alien species **(Figure 2.5)** suggesting that in general, more impacts on good quality of life have been reported where more alien species were found.

6. IPBES glossary: https://ipbes.net/glossary

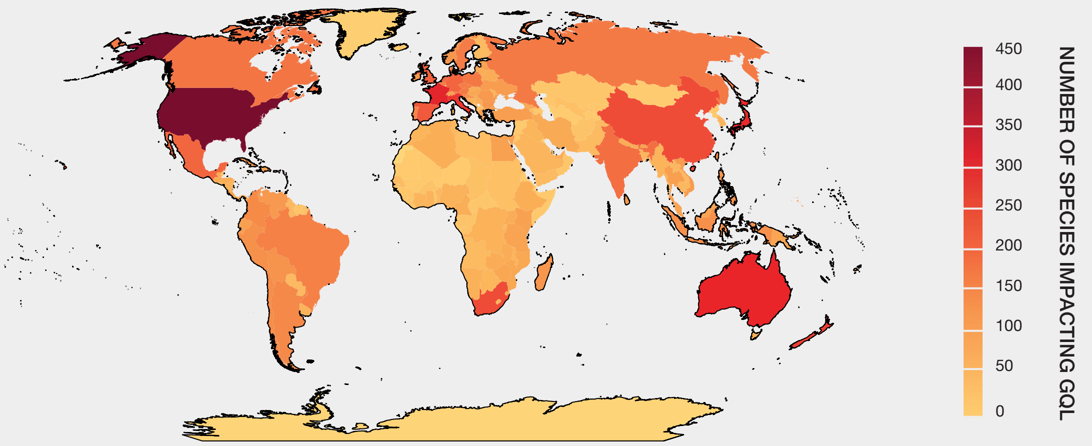

Figure 2 36 Map of invasive alien species numbers with reported impacts on good quality of life.

Species were identified through the literature review conducted by **Chapter 4** of this assessment (data management report available at: https:/doi.org/10.5281/zenodo.5766069) and the distributions of these species were extracted from the database used in **Chapter 2** (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

The total number of invasive alien species with impacts on good quality of life has risen continuously at a nearly linear rate since around 1830 **(Figure 2.37)**. During this time, the rate of increase remained relatively constant at around 15 new invasive alien species with impacts on good quality of life per five years (or three new species annually).

Most invasive alien species with impacts on good quality of life were insects (38 per cent), followed by vascular plants (29 per cent), fishes (7 per cent), molluscs (5 per cent), and mammals (5 per cent). Numerous widespread, well-known invasive alien species often negatively affect various aspects of good quality of life including culture, human health, and the local economy. High profile examples include fish species of the genus Oncorhynchus (trout and salmon) that have been introduced in many parts of the world (Crawford & Muir, 2008) and have changed local economies and livelihoods in areas. Such impacts include hybridization with native species and predation of native fishes (Kitano, 2004; Soto *et al.*, 2001; Woodford & Impson, 2004). The introduction of *Lates niloticus* (Nile perch) has changed the local socio-economic dynamics such as a decline in multi-fisheries subsistence and livelihood (Njiru *et al.*, 2018). In particular, women from marginalized communities have been disadvantaged by the effects of Lates niloticus on subsistence cichlid-based fisheries, and have had to adopt new livelihood practices, with prostitution being a primary one. This has, in turn, spurred inequality, social conflict, health issues (spread of human immunodeficiency virus (HIV) in particular), the loss of cultural practices, and reduced food

Box 2 11

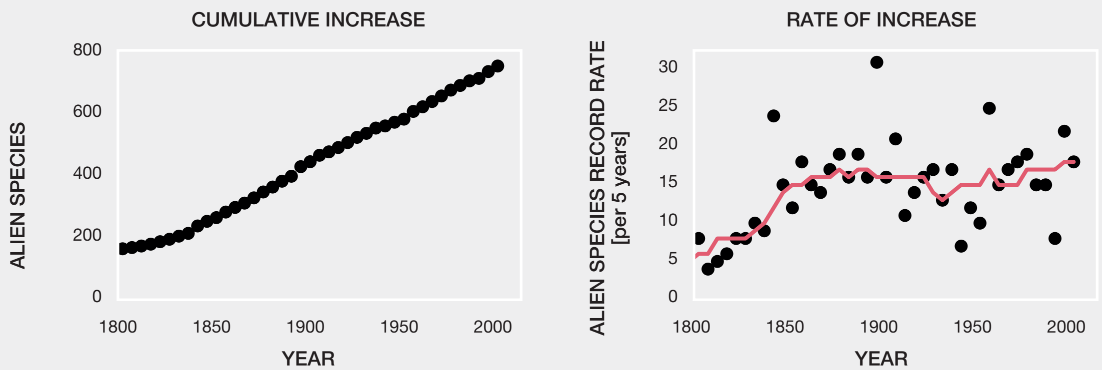

Figure 2 37 **Trends in numbers of invasive alien species with reported impacts on good quality of life.**

Trends are shown as cumulative numbers (left panel) and as rate of increase (i.e., numbers of species per five years) (right panel). The smoothed trend (line) is calculated as running median (**section 2.1.4** for further details about data sources and data processing). Species were identified through the literature review conducted by **Chapter 4** of this assessment (data management report available at: https:/doi.org/10.5281/zenodo.5766069) and the trends for these species were extracted from the database used in **Chapter 2** (**section 2.1.4** for further details about data sources and data processing). Note numbers presented may deviate from those reported in the text due to variation among data sources. A data management report for the data underlying this figure is available at https://doi.org/10.5281/zenodo.7615582

security for local communities, thus affecting human well-being (R. T. Shackleton *et al.*, 2018).

Another prominent example for an invasive alien species with impacts on good quality of life is Spodoptera frugiperda (fall armyworm). This alien insect pest has been spreading for decades and has wide-ranging impacts in many parts of the world including economic losses from reduced maize crop yields (Dassou *et al.*, 2021; De Groote *et al.*, 2020) and reduced local livelihood potential (Kassie *et al.*, 2020). The species is likely to spread further due to suitable climatic conditions (Day *et al.*, 2017; Early *et al.*, 2018). As another example, *Prosopis* spp. (mesquite) is one of the most widely distributed invasive tree species globally. These species have invaded many arid and semi-arid parts of the world, thereby reducing water available for humans and animals (Bekele *et al.*, 2018; Shiferaw *et al.*, 2021), impacting human health *via* allergies, asthma, and physical injuries (Al-Frayh *et al.*, 1999; Mwangi & Swallow, 2008), increasing malaria prevalence due to habitat provision (Muller *et al.*, 2017), reducing grazing capacity (S. Kumar & Mathur, 2014; Mwangi & Swallow, 2008; Ndhlovu *et al.*, 2011), and impacting local economies through increased management costs and loss of grazing (R. T. Shackleton *et al.*, 2014).

Focusing more specifically on Indigenous Peoples and local communities (i.e., typically ethnic groups who are descended from and identify with the original inhabitants of a given region; IPBES glossary[7]) and good quality of life, the assessment identified and assessed 131 regional case studies worldwide of the impacts of invasive alien species on the good quality of life and their effects for Indigenous Peoples and local communities. The most frequently reported species in the case studies were first identified, then species and their impacts on good quality of life concerning taxonomic groups, units of analyses, and IPBES regions. The findings suggested that the biggest impacts were from plant species (85 species, 65 per cent), of which most (79 species) were woody vascular plants.

The three most frequently reported invasive alien plants (38 cases) included either alone or in combination with other species were: *Lantana camara* (lantana), *Prosopis* spp., and *Chromolaena odorata* (Siam weed). Aquatic invasive alien plant species were reported in only six case studies. These included *Pontederia crassipes* (water hyacinth), *Phragmites australis* (common reed), *Hydrilla verticillate* (hydrilla), and *Cryptostegia grandiflora* (rubber vine), amongst others. Overall, fewer case studies (46 case studies) reported invasive alien species' impact on good quality of life for other taxonomic groups. These taxa included fish species (10 species) such as *Cyprinus carpio* (common carp), *Tilapia rendalli* (redbreast tilapia), *Oreochromis mossambicus* (Mozambique tilapia), and *Lates niloticus, Oncorhynchus mykiss* (rainbow trout). Insects (12 studies), were also reported including *Spodoptera frugiperda*, and *Agrilus planipennis* (emerald ash borer). Other taxa were not reported in any case studies.

The majority of case studies (60 per cent; 79 case studies) reported negative impacts of invasive alien species, while others reported both negative and positive impacts. Examples

7. IPBES glossary: https://ipbes.net/glossary

Box 2 11

include *Opuntia ficus-indica* (prickly pear), which is used for fodder and fence lines but has thorns that cause injury to humans and animals (S. E. Shackleton & Shackleton, 2018). Positive impacts of invasive alien species include feral pigs that provide meat (C. J. Robinson & Wallington, 2012), woody plants (e.g., *Acacia, Prosopis, Eucalyptus)* that provide biomass for compost, timber and wood charcoal production (Rogers *et al.*, 2017; Tassin *et al.*, 2012; B. W. van Wilgen, 2012), shade (S. E. Shackleton & Shackleton, 2018), products to sell (Tilahun *et al.*, 2017), and medicinal benefits (Witt *et al.*, 2019). Despite the benefits provided, the positive impacts of invasive alien species on good quality of life do not counteract their negative impacts.

Knowledge and data gaps

There were large differences in the number of studies from the different IPBES regions potentially representing knowledge and data gaps on the effects of invasive alien species on good quality of life. Asia and the Pacific had the most studies (54), followed by Africa (44), the Americas (28), and Europe and Central Asia (3). There appears to be a bias in case studies towards reporting the effects of invasive alien woody vascular plants (65 per cent) on good quality of life since there were many fewer case studies on other widespread alien species groups, particularly invertebrates, microbes, and mammals (5 per cent).

2.6 FUTURE DYNAMICS OF BIOLOGICAL INVASIONS

This section reports on the projected future dynamics of the trends and distribution of alien and invasive alien animal species in general (**section 2.6.1**), for animals (**section 2.6.2**), plants (**section 2.6.3**), and microorganisms (**section 2.6.4**), and addresses limitations for assessing future dynamics of biological invasions (**section 2.6.5**).

2.6.1 Overview of future dynamics of biological invasions

Recent increases in data availability and accessibility provide an improved baseline understanding of historic and current alien species richness and distributions that help to make new and improved projections (E. E. Dyer, Cassey, *et al.*, 2017; Pagad *et al.*, 2022; Seebens, Blackburn, *et al.*, 2017; van Kleunen *et al.*, 2019). However, many gaps still exist at the regional and taxonomic scales (Pyšek *et al.*, 2008). Approaches to forecast dynamics of biological invasions vary, including expert-based systems (e.g., based on individual experts in their field, Indigenous and local knowledge systems (**Glossary**), horizon scanning approaches), various modelling approaches (e.g., expert-based models, correlative models, process-based models, hybrid models; **Chapter 1**, **section 1.6.7.3**) or scenario approaches (exploratory scenarios, target-seeking scenarios, policy-screening scenarios; **Chapter 1**, **section 1.6.7.3**).

Generally, prediction and projection studies have been conducted from regional, continental to global scales (Bellard, Thuiller, *et al.*, 2013; Dullinger *et al.*, 2017) illustrating the potential current and future numbers and distribution of alien species. Studies cover one to multiple species within (e.g., cacti: Masocha & Dube, 2018; termites: Buczkowski & Bertelsmeier, 2017; ants: Bertelsmeier *et al.*, 2015, 2016; Fournier *et al.*, 2019) and across taxonomic groups (e.g., the 100 worst invaders globally as assessed by the IUCN ISSG: Bellard, Thuiller, *et al.*, 2013; Gallardo *et al.*, 2017).

On the global scale, quantitative projections of established alien species numbers under a business-as-usual scenario do exist for the period from 2005–2050 (Seebens, Bacher, *et al.*, 2021). For seven major taxonomic groups established alien species numbers are projected to increase across eight continental regions **(Figure 2.38)**. At the continental scale, the strongest relative increase in established alien species numbers of 64 per cent (2,543 ± 237 species) is expected for Europe, followed by temperate Asia (50 per cent; 1597 ± 197) and South America (49 per cent; 1,391 ± 258). Globally, an average relative increase of 36 per cent, equivalent to 1,195 ± 131 new established alien species is projected (Seebens, Bacher, *et al.*, 2021). A list of relative and absolute projected increases of established alien species numbers until 2050 is given in **Table 2.28**. However, given the projected acceleration of the majority of direct and indirect drivers of change in nature, it is likely that the numbers of established alien species will be higher than those predicted in the business-as-usual scenario **(Table 2.28)**. Comparing past and future trends, the rate of increase of established alien species numbers is expected to increase even further (i.e., acceleration) for arthropods and – to a lower degree – birds worldwide. In contrast, rates are projected to decline for mammals globally and partly for fishes, although rates are still positive, resulting in more alien species, but at a lower rate than observed before (Seebens, Bacher, *et al.*, 2021). However, the number of alien and invasive alien species is expected to rise even without the introduction of any new species by humans, because the majority of already established alien species are still spreading (Seebens, Blackburn, *et al.*, 2021). Thus, already established alien species are likely to spread further also to

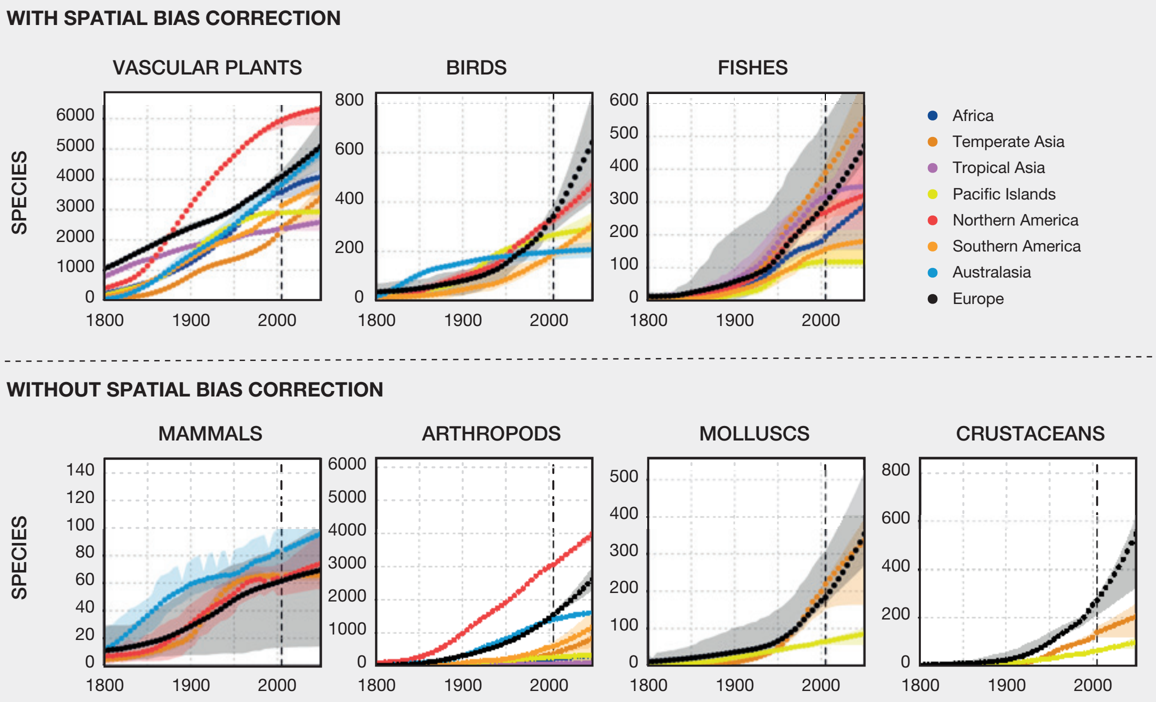

Figure 2 38 **Projected trends of established alien species numbers until 2050.**

Projections are shown for seven major taxonomic groups across eight global regions and based on a business-as-usual scenario that assumes that drivers facilitating biological invasions will develop in the future as has been observed during recent decades. For vascular plants, birds, and fishes a spatial bias correction was applied to account for spatial heterogeneity in data availability. This was not possible for the other taxonomic groups due to data deficiency. Trend lines show averaged trends out of repeated simulations, while variation around the means is indicated by shaded areas. From Seebens *et al.* (2021), https://doi.org/10.1111/gcb.15333, under license CC BY 4.0.

Table 2 28 **Projected relative (per cent) increases of established alien species numbers until 2050.**

Projections are representative for a business-as-usual scenario, assuming similar developments in drivers facilitating biological invasions as observed in the past. Values are mean estimates over 100 model runs with the upper and lower 2.5 per cent confidence interval given in square brackets. The absolute established alien species numbers increase averaged more than 100 model runs are provided in round brackets together with the standard deviation estimates. Data are from Seebens, Bacher, *et al.* (2021).

	Africa	Australasia	Europe	Northern America	Pacific Islands	South America	Temperate Asia	Tropical Asia
Mammals		14 [2, 29] (12±3)	13 [0, 167] (8±9)	16 [1, 46] (10±9)			0 [0, 10] (0±1)	
Birds	42 [0, 75] (59±26)	5 [1, 9] (9±4)	88 [44, 139] (299±53)	42 [32, 46] (138±11)	9 [1, 29] (24±22)	60 [10, 70] (115±20)		67 [36, 91] (78±15)
Fishes	49 [1, 75] (96±39)		59 [37, 104] (175±32)	20 [2, 70] (54±57)	0 [0, 1] (0±0)	16 [1, 96] (25±39)	42 [7, 62] (165±48)	10 [0, 76] (31±34)
Arthropods	51 [0, 73] (109±51)	15 [13, 18] (212±14)	69 [48, 85] (1072±92)	30 [24, 34] (927±31)	26 [1, 35] (70±17)	99 [0, 130] (582±249)	117 [57, 145] (445±87)	35 [0, 58] (24±13)
Molluscs			93 [59, 135] (170±31)		32 [2, 47] (21±7)		53 [3, 73] (116±40)	
Crustaceans			100 [51, 117] (273±34)		56 [10, 90] (36±8)		47 [0, 76] (66±18)	
Vascular plants	14 [4, 19] (503±113)	28 [22, 29] (1065±41)	24 [16, 39] (997±209)	6 [1, 7] (365±33)	1 [0, 2] (38±9)	21 [18, 25] (669±52)	41 [28, 54] (987±170)	10 [0, 17] (227±67)

neighbouring regions, which will result in further increases in alien species numbers regionally.

A literature review[8] on studies including models and scenarios of biological invasions shows that the current literature is dominated by correlative model approaches (57 per cent) and correlative scenarios (87 per cent) and that these studies mainly explore either long-term (2050-2100) or short-term (until 2030) trends (42 per cent and 30 per cent respectively) (**Chapter 1**, **section 1.6.7.3**).

The remainder of this section provides an overview of the general trends of predicted and projected alien species richness and distributions for different taxonomic groups and across scales.

2.6.2 Animals

For some bird species, such as *Corvus splendens* (house crow) and *Acridotheres tristis* (common myna), the current distributions indicate a large potential to spread to new areas (Magory Cohen *et al.*, 2019; Nyári *et al.*, 2006). Similarly, mammals such as *Sus scrofa* (feral pig), *Herpestes javanicus auropunctatus* (small Indian mongoose), and *Procyon lotor* (raccoon) often have a large potential of future invasions worldwide (Lewis *et al.*, 2017; Louppe *et al.*, 2019, 2020). In the marine realm, a study of 19 ascidian species finds a large invasion potential especially at higher latitudes (Lins *et al.*, 2018). For insects, several studies investigated the invasion potential of agricultural pest species (e.g., *Phthorimaea operculella* (potato tuber moth) (Kroschel *et al.*, 2013), *Bactrocera carambolae* (carambola fruit fly) (Marchioro, 2016), *Diabrotica* spp. (e.g., cucumber beetles) (Marchioro & Krechemer, 2018), *Bemisia tabaci* (tocacco whitefly) (Ramos *et al.*, 2018), *Spodoptera frugiperda* (fall armyworm) (Early *et al.*, 2018), *Halyomorpha halys* (brown marmorated stink bug) (Kriticos *et al.*, 2017), *Drosophila suzukii* (spotted wing drosophila) (L. A. dos Santos *et al.*, 2017)), and all studies found a high risk of invasion beyond the current realized distribution. Although less investigated, high invasion potentials have also been identified for other insect species (e.g., Fournier *et al.*, 2019; He *et al.*, 2012; H. Li *et al.*, 2006; Peacock & Worner, 2006). A study on the potential biological invasion risk of protected areas worldwide found that 95 per cent of the protected areas have high habitat suitability for alien mammal species across 11 taxonomic groups (X. Liu *et al.*, 2020).

An analysis of the 100 worst invaders of the world (as assessed by the IUCN ISSG) found a decreased potential for future global distribution of mammals, birds, fishes, reptiles, and amphibians, but an increase in distributions of aquatic and terrestrial invertebrates due to region specific projected changes in climate and land-use, using an ensemble species distribution models approach (Bellard, Leclerc, *et al.*, 2013). Other global and regional studies have focused on the future invasion potential for species from different taxonomic groups such as ants and termites (projected increases for 12 out of 13 species; e.g., Bertelsmeier *et al.*, 2013b, 2015; Buczkowski & Bertelsmeier, 2017; Y. Chen, 2008), beetles (projected increase; e.g., Berzitis *et al.*, 2014; Kistner-Thomas, 2019; C. Wang *et al.*, 2017), flies (northward shift and decrease in global suitability; e.g., Capinha *et al.*, 2014; M. P. Hill *et al.*, 2016; Qin, 2019; S. F. Ryan *et al.*, 2019), other insects (projected increase; e.g., M. P. Hill *et al.*, 2017; Lu *et al.*, 2020), amphibians (projected stable distribution or increase; e.g., Ficetola *et al.*, 2010; Forti *et al.*, 2017; Ihlow *et al.*, 2016), fish (projected increase; e.g., Dong *et al.*, 2020; Kramer *et al.*, 2017) and mammals (projected increase; e.g., Louppe *et al.*, 2019, 2020).

Under different scenarios of change of the global shipping network, which constitutes a major driver responsible for biological invasions (**Chapter 3**, **section 3.2.3.1**), and across taxonomic groups, high invasion risks have been identified for Asia and Europe (especially the Mediterranean) with a projected significant increase in the global invasion risk without management of shipping-mediated vectors (Sardain *et al.*, 2019). A risk assessment in the 19 Arctic ecoregions identified hotspots of future invasion for 23 invasive planktonic and benthic species in Hudson Bay, Northern Grand Banks/Labrador, Chukchi/Eastern Bering Seas and Barents/White Seas (Goldsmit *et al.*, 2020). Contrary to the projected Arctic expansion of the species their global projected range contracted, indicating a northward shift of future invasions (Goldsmit *et al.*, 2020). Mammal species, such as *Procyon lotor* (raccoon) and *Herpestes javanicus auropunctatus* (small Indian mongoose), are expected to shift to higher latitudes (Louppe *et al.*, 2019, 2020). Studies of individual fish species project potential future invasion risk across continents and at the regional scale (Dong *et al.*, 2020; Kramer *et al.*, 2017). For amphibians, two frog species (*Xenopus laevis* (African clawed frog) and *Lithobates catesbeianus* (American bullfrog)) are projected to have stable to decreasing future distributions under climate change (Ficetola *et al.*, 2010; Ihlow *et al.*, 2016). For insects, future potential distributions under climate change scenarios project poleward shifts (Capinha *et al.*, 2014; M. P. Hill *et al.*, 2016; Kistner-Thomas, 2019; Qin *et al.*, 2019) with many species increasing their potential distributions (Bellard, Thuiller, *et al.*, 2013; Bertelsmeier *et al.*, 2015; Buczkowski & Bertelsmeier, 2017; Y. Chen, 2008; Lu *et al.*, 2020; Qin *et al.*, 2019). At the same time, some insect species' distributions (e.g., *Aedes aegypti* (yellow fever mosquito), *Pheidole megacephala* (big-headed ant)) are projected to decrease as well, with the declines mainly located in tropical regions (Bertelsmeier *et al.*, 2013b; Capinha *et al.*, 2014; S. J. Ryan *et al.*, 2019).

8. Data management report available at: https://doi.org/10.5281/zenodo.5706520

In summary, the suite of studies available for projections of future dynamics of alien species suggests that overall ranges of alien species are expected to increase in most cases although with large variation due to a continuous introduction of new individuals and an expansion of ranges to other suitable habitats. In addition, ranges are expected to shift poleward because of global warming (Walther *et al.*, 2009). The total number of alien species is expected to increase until 2050 for most investigated taxonomic groups such as birds, fishes, mammals, arthropods, molluscs, and crustaceans (Seebens, Bacher, *et al.*, 2021). These trends are consistent across all continents except alien birds in Europe, alien mammals in tropical Asia, and alien fish on Pacific Islands, which are projected to reach a plateau. Relative increases between 2005 and 2050 range between 117 per cent (arthropods in temperate Asia) and 5 per cent (birds in Australasia) (Seebens, Bacher, *et al.*, 2021).

2.6.3 Plants

Potential hotspots of alien plants have been identified by modelling the distribution of individual plant species and projecting the distribution under future environmental conditions. For the 100 worst invaders (as defined by the IUCN), Europe, northern North America, and Oceania emerge as potential hotspots for invasion (Bellard *et al.*, 2016), while potential hotspots for cacti emerge in the Mediterranean, tropical savanna regions, and xeric shrubland biomes (Masocha & Dube, 2018). Other global studies on large sets of alien plant species identify high invasion risk in Europe, South America, North America, southwest China and New Zealand as well as the coast of West Africa and the southern coast of Asia (J.-Z. Wan *et al.*, 2016; Y. Wang & Xu, 2016). Regions of high invasion risk change depending on the taxa under investigation. For 10 parasitic Orobanchaceae species tropical and subtropical regions are most suitable for potential future invasions (Mohamed *et al.*, 2006). Higher potential future suitability has also been projected along roadsides (Azan *et al.*, 2015) and at the margins and buffer zones of protected areas (Gallardo *et al.*, 2017; Paclibar & Tadiosa, 2019), while potential future biological invasion risk is lower inside protected areas (Gallardo *et al.*, 2017; Paclibar & Tadiosa, 2019).

On the global scale, future distributions of some alien plant species are projected to expand (e.g., J.-Z. Wan *et al.*, 2016), while others will contract in parts of their current range (e.g., range contractions mainly at lower latitudes; Bellard, Leclerc, *et al.*, 2013) under different climate change scenarios. A recent study predicted the global distribution of 336 terrestrial invasive alien plants under future climate change scenarios (J.-Z. Wan *et al.*, 2016). It identifies the main future invasion hotspots for plant invasions to be in South America, Europe, New Zealand, and northern and Southern Africa (J.-Z. Wan *et al.*, 2016). Other studies focus either on single alien plant species (R. Ahmad *et al.*, 2019; Bourdôt *et al.*, 2012; Heshmati *et al.*, 2019) or sets of species within specific regions (e.g., Adams *et al.*, 2015; R. Ahmad *et al.*, 2019; J. M. Allen & Bradley, 2016; Dullinger *et al.*, 2017; Paclibar & Tadiosa, 2019). Most studies for Northern America and Europe report strong increases in overall potential future range sizes (e.g., Adhikari *et al.*, 2015; J. M. Allen & Bradley, 2016; Dullinger *et al.*, 2017) under global change, with the magnitude of change within these regions varying according to the species investigated and increases in suitable ranges are mainly directed towards higher latitudes (J. M. Allen & Bradley, 2016). Studies for the United States and Europe project that most current invasion hotspots will remain stable spatially, but potential invasion alien species richness will increase between 64 to 102 per cent (J. M. Allen & Bradley, 2016; Dullinger *et al.*, 2017).

For Europe, a prediction of future development of plant invasions until 2080 under three socioeconomic scenarios differing in focus on economic growth vs. sustainability has been made based on data from vegetation plots (Chytrý *et al.*, 2012). Under all scenarios an increase in the level of invasion was projected for north-western and northern Europe, and under two of the scenarios a decrease for some agricultural areas of Eastern Europe where abandonment of agricultural land is expected. However, the implementation of sustainability policies would not automatically restrict the spread of alien plants (Chytrý *et al.*, 2012).

Following a business-as-usual scenario, thereby assuming that drivers will develop in the future as observed in the past, alien vascular plants species numbers are expected to increase steadily across all continents with only North America showing a weak sign of saturation by 2050 (Seebens, Bacher, *et al.*, 2021; **Figure 2.38**). The range of the projected increase of alien vascular plants lies between 1 per cent (Pacific Islands) and 41 per cent (Temperate Asia) from 2005-2050 **(Table 2.28)**. Likewise, relative increases in species numbers are projected to increase more strongly in aquatic than non-aquatic environments (Seebens, Bacher, *et al.*, 2021). In the marine realm, future increases in alien algae species introductions are projected for Asia and Europe (Seebens, Bacher, *et al.*, 2021) and mainly along the major shipping routes (Sardain *et al.*, 2019).

2.6.4 Microorganisms

A recent review of species distribution models used for fungi has identified 75 studies predicting the potential distribution of fungi under current climates (Hao *et al.*, 2020). The majority of studies deal with one species only or with multiple species from the same genus (e.g., *Phytophthora*; Scott *et al.*, 2019) and generally invasion risk is predicted to be higher as currently observed, both in terms of numbers of alien fungi present (Barwell *et al.*, 2021; Bebber *et al.*, 2019; Scott *et al.*, 2019) and of occupied range (e.g., Feldmeier *et al.*, 2016; Kriticos *et al.*, 2013; Yonow *et al.*, 2013). For crop pests including herbivorous arthropods, pathogenic microbes, and virus species numbers within regions are predicted to be higher than observed levels (Bebber *et al.*, 2019) and hotspots of pest invasion are located in Mesoamerica, Europe, North-East Asia and Australia (Bebber, 2015).

Global plant pathogen studies project an increase in potentially suitable areas, especially towards higher latitudes (Avila *et al.*, 2019; Burgess *et al.*, 2017). While for some pathogens (e.g., *Phytophthora cinnamomi* (Phytophthora dieback); Burgess *et al.*, 2017) the entire potential future environmental range is modelled, other approaches couple both the pathogens and hosts when modelling future ranges (e.g., *Diuraphis noxia* (Russian wheat aphid), Avila *et al.*, 2019). Additionally, there are approaches that extend distributional invasion risk measures by impact assessments that assess the overlap of the potential future distribution and cropland area (e.g., *Raoiella indica* (red palm mite); Amaro & de Morais, 2013). Pathogen distribution in many cases is linked to introduced invasive alien species that act as host species and projected invasions thus are inferred from host species presence and distribution change (e.g., chytridiomycosis; O'Hanlon *et al.*, 2018). Crop pests are projected to shift poleward under climate change and increased human activities (Bebber *et al.*, 2013; Fisher *et al.*, 2012, 2020) and under current observed trends the main crop producing countries will be saturated with crop pathogens by 2050 (Bebber *et al.*, 2014). In the marine realm, projections of planktonic and benthic species, as well as algae, identify a future potential poleward shift of alien species under climate change scenarios (Goldsmit *et al.*, 2020; Seebens *et al.*, 2016).

2.6.5 Limitations for assessing future dynamics

Projections of future dynamics of alien and invasive alien species are severely limited by 1) data availability of past and current distributions of species, 2) knowledge gaps of the past and current distribution of species, 3) knowledge gaps of the understanding of causal relationships between species occurrences, environmental changes, drivers of change in nature, biological invasions, and impacts caused by invasive alien species, 4) lack of models to robustly predict future dynamics of biological invasions, and 5) the lack of scenarios covering a range of plausible future dynamics of drivers of change, which would allow exploring future trends under different scenarios. While models and scenarios can still be further developed, closing data gaps, particularly of historic distributions, is very difficult and even impossible in many cases.

Most global studies focus on either individual species or different subsets of species based on specific characteristics (e.g., the 100 of the worst global invaders as assessed by the IUCN ISSG; Bellard, Thuiller, *et al.*, 2013; Gallardo *et al.*, 2017) or on technical criteria such as data availability. Consequently, it is difficult to discern a comprehensive pattern of potential future alien species richness and distribution for individual taxonomic groups (but see Seebens, Bacher, *et al.*, 2021). Additionally, information on alien species distributions is not spatially and taxonomically homogeneous and is biased towards specific regions of the world, like Europe and Northern America (A. C. Hughes *et al.*, 2021; C. Meyer *et al.*, 2016). Although online portals for storing biodiversity data such as GBIF provide billions of occurrence records, the data still covers just a fraction of known species. This limitation in accessibility to species occurrence data severely hampers modelling approaches for predicting and projecting future alien species richness and distribution patterns (**Chapter 1, section 1.6.7.3**).

A major challenge for most groups of microorganisms and fungi is the delineation of their native range resulting from a lack of data for these groups in general, as well as from high taxonomic uncertainty due to frequent historic changes and adaptations of the taxonomic concepts (e.g., due to new technological advancements; De Clerck *et al.*, 2013; Essl *et al.*, 2018; Hao *et al.*, 2020; Sharma *et al.*, 2015). In the absence of the ability to distinguish between the native and alien range of a species, robust risk assessments and predictions on the potential future spread and distribution are not possible.

In addition, alien pathogen research largely focusses on human pathogens, livestock, and cultivated plants, neglecting other facets of biodiversity and ecosystem services (Fischer *et al.*, 2012; Peeler *et al.*, 2011; Roy *et al.*, 2017; Usher, 1986). Further, most invasive alien pathogens are only described once their impacts are recognized in the invaded range (Roy *et al.*, 2017) hampering the identification of potential future alien species risk assessments. Finally, many pathogens undergo host shifts in the invaded range (McTaggart *et al.*, 2016; Peeler *et al.*, 2011; Roy *et al.*, 2017), which can strongly affect disease-induced host mortality in the invaded range, which increases with the evolutionary distance between the native and alien host species (Farrell & Davies, 2019). Such information of host-pathogen associations and interaction however are skewed to few well-studied alien pathogens (Farrell & Davies, 2019).

The systematic literature review of the models and scenarios[9] revealed distinct trends and research gaps. Research is mainly focused on the Americas, followed by Europe and Central Asia, and Asia and the Pacific,

9. Data management report available at: https://doi.org/10.5281/zenodo.5706520

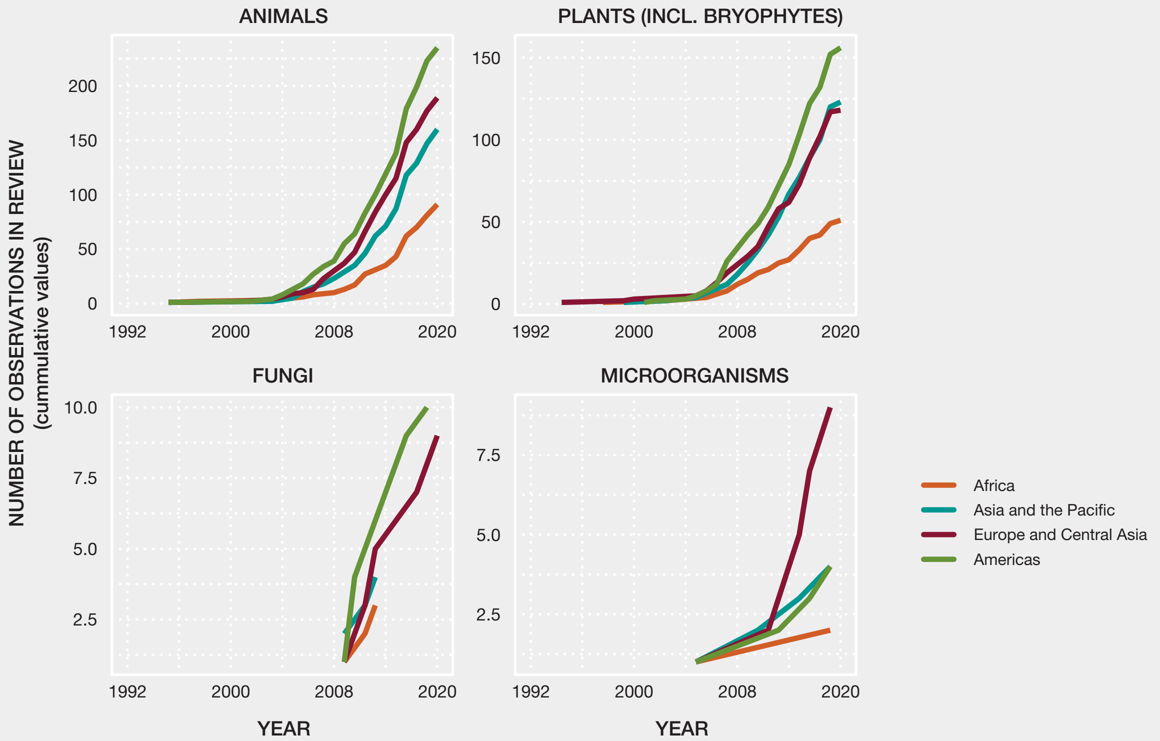

Figure 2 39 **Trends in the number of analyses of future projections across IPBES regions.**

Number of observations is not equal to the number of studies as studies can report on more than one IPBES region and/or taxonomic group. The time reported on the y-axis refers to the date of publication of the respective study. Values are based on a comprehensive literature review about scenarios and models of biological invasions, a data management report is available at https://doi.org/10.5281/zenodo.5706520

indicating a large knowledge gap in models and scenario studies for Africa. The number of studies is accelerating at an equal pace across IPBES regions **(Figure 2.39)**. Plants and animal studies are the most studied taxonomic groups; however, when further separating animals into finer classes, it is clear that animal studies are dominated by research on invertebrates and overall plants are the predominantly studied group, which is consistent over time. Studies for fungi and microorganisms are lacking (**Chapter 1**, **section 1.6.7.3**). Studies projecting alien species distributions into the future are largely lacking for the marine realm and also not very numerous for freshwater regions compared to the terrestrial realm. While the number of studies has accelerated over time, it is more prominent in the terrestrial realm and especially in the Americas **(Figure 2.40)**. Finally, most scenario projections explore long-term (2050-2100) and short-term (until 2030) trends. Very few studies follow a backcasting approach that involves setting a desirable future end-point and determining possible pathways including policy measures to reach that end-point (Dreborg, 1996).

To summarize, there is a distinct lack of model and scenario studies for Africa and Asia and the Pacific, the marine and freshwater realms. Finally, the scientific literature is dominated by correlative models whose application has increased more rapidly than for other modelling approaches. Also, process-based models have accelerated in their application; however, the application of hybrid models that combine both correlative and process-based approaches is not very common. Expert-based systems are not utilized for model and scenario studies implying a major gap in the utilization of these knowledge systems. A comprehensive overview of the review can be found in **Chapter 1**, **section 1.6.7.3** and on identified gaps in **Chapter 6**, **Table 6.10** and **section 6.6.1.1** and all information and data are available in the data management report.[10]

Finally, in addition to data and knowledge gaps, the prediction of future dynamics of biological invasions is severely impeded by a lack of models to predict those dynamics and by scenarios to explore variations among plausible futures. Although several modelling approaches exist for individual species, regions, or drivers as presented

10. Data management report available at: https://doi.org/10.5281/zenodo.5706520

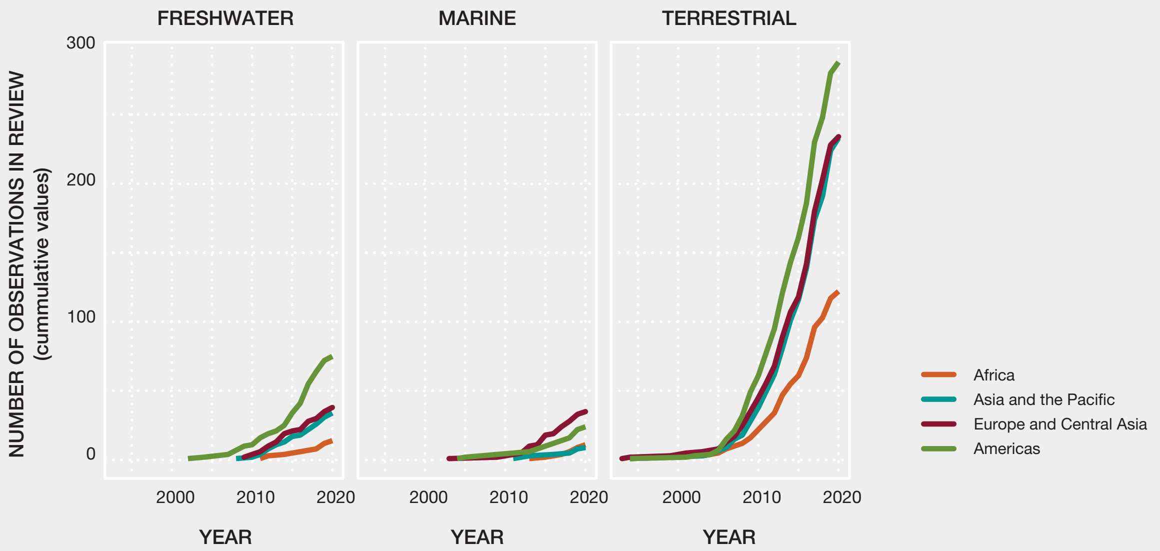

Figure 2 40 **Trends in the number of analyses of future projections for realms.**

Realms are defined as freshwater, marine and terrestrial. Number of observations is not equal to the number of studies as studies can report on more than one IPBES region and realm. The time reported on the y-axis refers to the date of publication of the respective study. Values are based on a comprehensive literature review about scenarios and models of biological invasions, a data management report is available at https://doi.org/10.5281/zenodo.5706520

above, no models are available to simulate biological invasions at large spatial and temporal scales, including a range of different species, drivers and impacts. In addition, quantitative scenarios of biological invasions are missing, which hampers the prediction of biological invasions under different plausible futures of driver developments. Qualitative scenario description recently became available (Roura-Pascual *et al.*, 2021), but the quantification and applications in modelling exercises remain to be tested. The field of biological invasions is distinctly lagging behind the progress of other drivers of change in nature, such as climate change and land-use changes, where much more attention has been paid over recent decades to develop models and scenarios.

2.7 CONCLUSIONS

The main objective of this chapter was to provide a global overview of the current understanding of the temporal trends and the spatial distribution (i.e., status) of alien and invasive alien species. By conducting extensive literature reviews and consulting experts from all over the world, assessment experts have gathered information on the trends and status of alien and invasive alien species across a wide range of taxonomic groups, geographic regions, and ecosystems. This assessment strove to provide an overview, which is as balanced as possible in terms of geographic and taxonomic coverage of species. However, complete coverage across all taxa, habitats, and regions is not possible due to many data and knowledge gaps. In some cases, the widespread gaps make a truly global and extensive assessment of the trends and status difficult. In addition, even well-sampled taxa and regions likely have incomplete information. Although this assessment considered a huge number of publications, including scientific publications, reports, and books in various languages, and consulted many experts, many sources of information could not be considered in this chapter, particularly non-English publications and grey literature, which are difficult to access if experts from that field or region are not directly involved.

Although this chapter provides the most comprehensive assessment of the trends and status of the distribution of known alien and invasive alien species, it is nonetheless based on incomplete data, the extent of which varies by taxa, region, and habitat. However, the existence of such gaps does not imply that any robust conclusions cannot be drawn. In fact, there is a good understanding of the trends and status of alien species for many taxonomic groups and regions, which are presented in this chapter, and the most robust and general conclusions are shown in the executive summary at the beginning of this chapter. However, with incomplete data it is necessary to verify available information by assessing trends and status based on scientific expertise and taking underlying biases into account.

Biological invasions are complex and intertwined with human transportation and goods, as well as other components of global change such as land use change, climate change, and human disturbances. This ecological complexity, the diversity and abundance of alien species, and the difficulty of identifying invaders in new environments, make their prevention and management challenging. The data presented in this chapter demonstrate that there is almost no place on Earth that has not experienced alien species introductions. It also shows that alien species introductions to new ranges are increasing across all taxa, all IPBES regions, and all units of analysis and that there are large data and knowledge gaps across these three sectors. The immediate result is that biological invasions are underestimated, with many species not yet identified as invasive and many ecosystems not yet recorded as invaded, or invaded by all the alien species that are present.

Decision makers often interpret research and develop policies to address biological invasions based on incomplete and biased data. Identifying and closing these data and knowledge gaps is essential to assess and address biological invasions more accurately and comprehensively. While gathering the information underlying this chapter, experts have identified the following major limitations which hindered the assessment:

1. **Lack of regional alien species lists:** For many taxonomic groups, particularly among invertebrates and microorganisms, lists of reported alien species are lacking for many countries. Even for ecologically and economically important groups such as insects, such lists are often lacking.

2. **Incomplete data:** Available lists of alien species occurrences are often incomplete or outdated. While difficult to identify, a comparison of alien species numbers across countries often revealed strong differences among neighbouring countries, differences that are likely influenced by degree of survey intensity rather than actual occurrences. In addition, the spread of alien species is highly dynamic and thus maintaining an up-to-date list of alien species occurrences requires regular monitoring which is rare. Even more rare are data on the abundances of individual populations. They are so scarce that experts were unable to consider alien species population abundances in this chapter.

3. **Lack of standardization:** Available lists of alien species were often generated using different terms that vary in their definitions, concepts (including taxonomies), and data collection and sampling practices, making comparisons of available information across regions and taxa difficult. This is particularly problematic for distinguishing a species' invasion status such as introduced, established, and invasive; these distinctions are often not specified, and if they are, the applied definitions are often not provided. Ideally, data is reported using standard concepts and terminologies, which are also explicitly detailed in the description of the data.

4. **Coarse spatial resolution:** The information on alien species occurrences is usually provided only at a coarse spatial resolution, such as the country level. However, the distribution of alien species within a country is often aggregated towards certain geographic areas within national borders. For a thorough assessment of biological invasions across spatial scales, it is essential to obtain information at finer resolutions that are ideally associated with coordinates of alien species occurrences.

Closing these gaps poses huge challenges to the scientific community. Below is a list of a few key challenges to improving assessments of the trends and status of alien and invasive alien species.

Improving collaboration

To fill data gaps and make invasion science truly global, greater, and more equitable, international collaboration is needed to build more global networks for monitoring, data sharing, and technology transfer (Kuebbing *et al.*, 2022; Meyerson *et al.*, 2022; Nuñez *et al.*, 2021; Packer *et al.*, 2017). The trend towards open-source software, such as QGIS and statistical environments such as R, is helping to reduce disparities between rich and poorer regions, but costs associated with training scientists and executing research as well as prohibitive journal publication costs present serious obstacles (**Chapter 6**, **section 6.6.2.4**). Many invasive alien species-focused research networks, database repositories, intergovernmental and international organizations, and international agreements are already in place (reviewed in Meyerson *et al.*, 2022). Despite these efforts, additional coordination and collaboration are needed, particularly because individual countries often do not have the capacities to respond to the issues of biological invasions sufficiently (**Chapter 6**, **section 6.3.1.1**; Early *et al.*, 2016; Pyšek, Hulme, *et al.*, 2020). In addition, it would be beneficial to engage in a two- or multi-way discussion with public and stakeholders through a new "dialogue communication model" or "public engagement model" (**Chapter 5**, **section 5.2.1**; **Chapter 6**, **section 6.4**), based on a genuine interchange with the public that recognizes and incorporates differences in knowledge, values, perspectives, and interests (Courchamp *et al.*, 2017). This will allow better understanding of biological invasions and supporting data acquisition, research and management.

Closing knowledge gaps

Thoroughly assessing the trends and status of biodiversity requires deep knowledge about nature and the ecosystems supporting biodiversity. Without knowing the species and their life histories, their interactions, and the mechanisms shaping environments worldwide, the state of biodiversity cannot be fully assessed. While information about nature is accumulating at an unprecedented pace, there are still major knowledge gaps, particularly for relatively inconspicuous organisms such as invertebrates, fungi, and microorganisms, and less accessible systems such as in marine habitats, but also inland waters, and in geographic areas such as Central Africa, Central Asia, and remote islands. In addition, there is a lack of an adequate understanding of biotic and abiotic species interactions, without which experts cannot fully grasp how species respond to environmental changes nor build models predicting future biodiversity change under different scenarios of human development. Closing these knowledge gaps is therefore essential to fully inform policies that can safeguard nature and move societies towards sustainability.

Efficient and standardized sampling and data processing

Comprehensive and thorough assessments of biological invasions and biodiversity in general need global and comprehensive monitoring and databases (Latombe *et al.*, 2017; Meyerson *et al.*, 2022; Packer *et al.*, 2017), which can only be obtained by implementing the following:

- Collection of records of alien species occurrences, and regular and repeated deposition into publicly accessible databases, particularly in regions and for taxonomic groups with the most severe gaps.
- Mobilization of existing data by making it accessible to the wider community in electronic formats and by providing these data under the Findable, Accessible, Interoperable, Reusable (FAIR) principles of open science (Wilkinson *et al.*, 2016).
- Standardization of available and accessible data to allow comparison, which could be accomplished by adopting a standard terminology for biodiversity information as Darwin Core has done, and by using open and widely used data formats such as csv or txt (Groom *et al.*, 2019).
- Documentation of data transformation steps, ensuring that they are repeatable and associated with the data (Seebens *et al.*, 2020).
- Finally, integration of standardized data into open databases or data portals such as GBIF or the Ocean Biodiversity Information System (OBIS) to enable researchers and stakeholders to conduct tailored biodiversity assessments.

Ideally, all steps from recording to storing data would follow standard and published protocols to make science, decision-making, and the assessment of biodiversity comprehensive, transparent, interoperable and reproducible, which ultimately increases trust in results and decisions (e.g., De Pooter *et al.*, 2017; Groom *et al.*, 2017; Haider *et al.*, 2022; Roy *et al.*, 2018).

Technological advances

Similar to the increase in information, technologies are developing rapidly including those designed to monitor biodiversity. Advances range from new satellite products to environmental DNA to fully automated biodiversity measurement stations. For example, satellites now provide opportunities to measure not only vegetation patterns at high resolution but also to track the movement of species or to distinguish individual plant species and measure plant traits which can provide early detection of new alien species introductions. Likewise, environmental DNA can help to populate lists of species occurring in certain areas, including rare species and emerging new alien species. Cameras and pattern recognition through artificial intelligence can identify species at comparatively low cost but on large geographic scales. Drones can now monitor biodiversity and fully automated biodiversity stations similar to weather stations are currently developed to obtain high resolution recordings of biodiversity. However, although these developments are promising, the technologies often still require major advancements to get ready for measuring biodiversity at the species level. In addition, many technological solutions are still used in isolation and large-scale solutions to obtain comprehensive coverage of biodiversity monitoring have not yet been achieved.

Engagement with policy makers

Progress towards addressing data gaps for biological invasions can benefit from engagement by policy makers, funding, trained (citizen) scientists, and technicians, adequate infrastructure to achieve standardized tools for long-term monitoring, modular regulatory frameworks that integrate incentives and compliance mechanisms with respect for diverse transcultural needs, biosecurity awareness and measures and synergies with other conservation strategies (Meyerson *et al.*, 2022; **Chapter 5**, **section 5.4.3.2(a)**; **Chapter 6**, **section 6.6.2.1**).

Inclusive biodiversity monitoring (citizen science, Indigenous Peoples and local communities)

Global comprehensive taxonomic monitoring of alien and native biota could be improved through engagement with people outside of academia, agencies, and institutions. People interested in nature and willing to contribute to recording of species occurrences could be encouraged to provide their knowledge and findings to other people and databases through, for example, community science projects, participatory research programmes and online platforms such as iNaturalist, CoralWatch, Project Noah, or e-Bird (Aristeidou *et al.*, 2021; Ballard, Dixon, *et al.*, 2017; Ballard, Robinson, *et al.*, 2017; McKinley *et al.*, 2017; **Chapter 1**, **Box 1.15**; **Chapter 6**, **section 6.6.2.1**). Such a large scale, ideally global, data reporting and sharing programme requires, however, concerted efforts of the international community and thus would benefit from greater efforts and incentives by governments and institutions to encourage people to contribute. Obtaining data through community science of sufficient quality for use in biodiversity assessments can be achieved through concerted coordination and organization, training, guidance, and funding. Standards for sampling and reporting have to be defined and adhered to, and needs and goals must consider the requirements of individual communities. In this way, inclusive biodiversity monitoring would include Indigenous Peoples and local communities who have a deep understanding about those areas that are least represented in global biodiversity assessments. Such an approach to fill data gaps for alien and invasive alien species is inclusive, adaptive, and flexible. As integrated and collaborative networks develop, effective global strategies to address invasive alien species will finally be met.

Accounting for incomplete knowledge

Several data gaps could be filled by increasing efforts and investments into biodiversity research and monitoring. However, it seems unlikely that obtaining complete and regular data at large geographic scales is achievable. Thus, it is also necessary to not only acknowledge the lack of information, but to also quantify uncertainty and incompleteness of data and to explicitly account for those biases in biodiversity assessments and analyses. This requires the development and adoption of standardized methods to quantify uncertainty. Having a standardized approach to measure and account for incomplete data would increase robustness of the results, and increase confidence in individual reports of biological invasions and biodiversity research more generally (Franz & Sterner, 2018).

REFERENCES

Abbott, I., Marchant, N., & Cranfield, R. (2000). Long-term change in the floristic composition and vegetation structure of Carnac Island, Western Australia. *Journal of Biogeography, 27*(2), 333–346. https://doi.org/10.1046/j.1365-2699.2000.00409.x

Abella, S., Fisichelli, N. A., Schmid, S. M., Embrey, T. M., Hughson, D. L., & Cipra, J. (2015). Status and management of non-native plant invasion in three of the largest national parks in the United States. *Nature Conservation, 10*, 71–94. https://doi.org/10.3897/natureconservation.10.4407

Abellan, P., Carrete, M., Anadon, J. D., Cardador, L., & Tella, J. L. (2016). Non-random patterns and temporal trends (1912-2012) in the transport, introduction and establishment of exotic birds in Spain and Portugal. *Diversity and Distributions, 22*(3), 263–273. https://doi.org/10.1111/ddi.12403

Acevedo-Rodríguez, P., & Strong, M. T. (2008). Floristic richness and affinities in the West Indies. *The Botanical Review, 74*(1), 5–36. https://doi.org/10.1007/s12229-008-9000-1

Ackerman, J. D., Tremblay, R. L., Rojas-Sandoval, J., & Hernández-Figueroa, E. (2017). Biotic resistance in the tropics: Patterns of seed plant invasions within an island. *Biological Invasions, 19*(1), 315–328. https://doi.org/10.1007/s10530-016-1281-4

Acosta, F., Zamor, R. M., Najar, F. Z., Roe, B. A., & Hambright, K. D. (2015). Dynamics of an experimental microbial invasion. *Proceedings of the National Academy of Sciences of the United States of America, 112*(37), 11594–11599. https://doi.org/10.1073/pnas.1505204112

Adams, V. M., Petty, A. M., Douglas, M. M., Buckley, Y. M., Ferdinands, K. B., Okazaki, T., Ko, D. W., & Setterfield, S. A. (2015). Distribution, demography and dispersal model of spatial spread of invasive plant populations with limited data. *Methods in Ecology and Evolution, 6*(7), 782–794. https://doi.org/10.1111/2041-210X.12392

ADB. (2012). *The state of Pacific towns and cities: Urbanization in ADB's Pacific developing member countries.* Asian Development Bank. http://hdl.handle.net/11540/868

Addison, J. A. (2009). Distribution and impacts of invasive earthworms in Canadian forest ecosystems. *Biological Invasions, 11*(1), 59–79. https://doi.org/10.1007/s10530-008-9320-4

Adhikari, D., Tiwary, R., & Barik, S. K. (2015). Modelling hotspots for invasive alien plants in India. *PLoS ONE, 10*(7), e0134665. https://doi.org/10.1371/journal.pone.0134665

Agostinho, A. A., Thomaz, S. M., & Gomes, L. C. (2005). Conservation of the biodiversity of Brazil's inland waters. *Conservation Biology, 19*(3), 646–652. https://doi.org/10.1111/j.1523-1739.2005.00701.x

Ahmad, M., Uniyal, S. K., & Singh, R. D. (2018). Patterns of alien plant species richness across gradients of altitude: Analyses from the Himalayan state of Himachal Pradesh. *Tropical Ecology, 59.*

Ahmad, R., Khuroo, A. A., Hamid, M., Charles, B., & Rashid, I. (2019). Predicting invasion potential and niche dynamics of *Parthenium hysterophorus* (Congress grass) in India under projected climate change. *Biodiversity and Conservation, 28*(8–9), 2319–2344. https://doi.org/10.1007/s10531-019-01775-y

Akbulut, S., & Karaköse, M. (2018). Some Natural and Exotic Invasive Plant Species in Turkey. In I. Christova, E. Strauss, A.-A. Gad, & I. Curebal (Eds.), *Science, Ecology and Engineering Research inthe Globalizing World* (p. 530). St. Kliment Ohridski University Press.

Akhtar, J., Chalam, V. C., Kumar, P., Kiran, R., & Dubey, S. C. (2019). Plant quarantine—A phytosanitary requirement for disease free import of plant genetic resources in India. In J. Stanley, K. K. Mishra, A. R. N. S. Subbanna, H. Rajashekara, & A. Pattnayak (Eds.), *Integrated pest management in major crops* (pp. 153–163). ICAR-Vivekananda Parvatiya Krishi Anusandhan Sansthan, Almora.

Akhtar, J., Gupta, K., Gawade, B. H., Kumar, P., Meena, B. R., Kiran, R., & Chalam, V. C. (2021). Strategies to combat the threat of quarantine pests to the plant health and food security. In N. K. Bharat & H. R. Guatam (Eds.), *Technology strides in plant health management* (pp. 273–288). Neoti Book Agency Pvt. Ltd. https://www.researchgate.net/profile/Pardeep-Kumar-51/publication/352537551_Strategies_to_combat_the_threat_of_quarantine_pests_to_the_plant_health_and_food_safety/links/60cd88e4458515dc1791d5e5/Strategies-to-combat-the-threat-of-quarantine-pests-to-the-plant-health-and-food-safety.pdf

Alexander, J. M. (2010). Genetic differences in the elevational limits of native and introduced *Lactuca serriola* populations. *Journal of Biogeography, 37*(10), 1951–1961. https://doi.org/10.1111/j.1365-2699.2010.02335.x

Alexander, J. M., Kueffer, C., Daehler, C. C., Edwards, P. J., Pauchard, A., Seipel, T., & MIREN Consortium. (2011). Assembly of nonnative floras along elevational gradients explained by directional ecological filtering. *Proceedings of the National Academy of Sciences, 108*(2), 656–661. https://doi.org/10.1073/pnas.1013136108

Alexander, J. M., Lembrechts, J. J., Cavieres, L. A., Daehler, C., Haider, S., Kueffer, C., Liu, G., McDougall, K., Milbau, A., Pauchard, A., Rew, L. J., & Seipel, T. (2016). Plant invasions into mountains and alpine ecosystems: Current status and future challenges. *Alpine Botany, 126*(2), 89–103. https://doi.org/10.1007/s00035-016-0172-8

Al-Frayh, A., Hasnain, S. M., Gad-el-Rab, M. O., Al-Turki, T., Al-Mobeireek, K., & Al-Sedairy, S. T. (1999). Human Sensitization to *Prosopis juliflora* Antigen in Saudi Arabia. *Annals of Saudi Medicine, 19*(4), 331–336. https://doi.org/10.5144/0256-4947.1999.331

Alidoost Salimi, P., Creed, J. C., Esch, M. M., Fenner, D., Jaafar, Z., Levesque, J. C., Montgomery, A. D., Alidoost Salimi, M., Edward, J. K. P., Raj, K. D., & Sweet, M. (2021). A review of the diversity and impact of invasive non-native species in tropical marine ecosystems. *Marine Biodiversity Records, 14*(1), 11. https://doi.org/10.1186/s41200-021-00206-8

Allen, E., Noseworthy, M., & Ormsby, M. (2017). Phytosanitary measures to reduce the movement of forest pests with the international trade of wood products. *Biological Invasions, 19*(11), 3365–3376. https://doi.org/10.1007/s10530-017-1515-0

Allen, J. A., Brown, C. S., & Stohlgren, T. J. (2009). Non-native plant invasions of United States National Parks. *Biological*

Invasions, 11(10), 2195–2207. https://doi.org/10.1007/s10530-008-9376-1

Allen, J. M., & Bradley, B. A. (2016). Out of the weeds? Reduced plant invasion risk with climate change in the continental United States. *Biological Conservation, 203*, 306–312. https://doi.org/10.1016/j.biocon.2016.09.015

Allen, W. L., Street, S. E., & Capellini, I. (2017). Fast life history traits promote invasion success in amphibians and reptiles. *Ecology Letters, 20*(2), 222–230. https://doi.org/10.1111/ele.12728

Almeida, J. (2018). New additions to the exotic vascular flora of continental Portugal. *Flora Mediterranea, 28*, 259–278. https://doi.org/10.7320/FlMedit28.259

Almeida, J., & Freitas, H. (2001). The exotic and invasive flora of Portugal. *Botanica Complutensis, 25*, 317–327.

Amaro, G., & de Morais, E. G. F. (2013). Potential geographical distribution of the red palm mite in South America. In *Experimental and Applied Acarology* (3; Vol. 60, Issue 3, pp. 343–355). SPRINGER. https://doi.org/10.1007/s10493-012-9651-9

Anderson, P. K., Cunningham, A. A., Patel, N. G., Morales, F. J., Epstein, P. R., & Daszak, P. (2004). Emerging infectious diseases of plants: Pathogen pollution, climate change and agrotechnology drivers. *Trends in Ecology & Evolution, 19*(10), 535–544. https://doi.org/10.1016/j.tree.2004.07.021

Andradi-Brown, D. A. (2019). Invasive Lionfish (*Pterois volitans* and *P. miles*): Distribution, Impact, and Management. In Y. Loya, K. A. Puglise, & T. C. L. Bridge (Eds.), *Mesophotic Coral Ecosystems* (pp. 931–941). Springer International Publishing. https://doi.org/10.1007/978-3-319-92735-0_48

Andreas, B. K. (1989). *The vascular flora of the glaciated Allegheny Plateau region of Ohio.* College of Biological Sciences, Ohio State University.

Andriaholinirina, N., Baden, A., Blanco, M., Chikhi, L., Cooke, A., Davies, N., Dolch, R., Donati, G., Ganzhorn, J., Golden, C., Groeneveld, L. F., Hapke, A., Irwin, M., Johnson, S., Kappeler, P., King, T., Lewis, R., Louis, E. E., Markolf, M., … Zaramody, A. (2014a). *Daubentonia madagascariensis: A.: The IUCN Red List of Threatened Species 2014: e.T6302A16114609* [dataset]. International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.UK.2014-1.RLTS.T6302A16114609.en

Andriaholinirina, N., Baden, A., Blanco, M., Chikhi, L., Cooke, A., Davies, N., Dolch, R., Donati, G., Ganzhorn, J., Golden, C., Groeneveld, L. F., Hapke, A., Irwin, M., Johnson, S., Kappeler, P., King, T., Lewis, R., Louis, E. E., Markolf, M., … Zaramody, A. (2014b). *Varecia variegata: A.: The IUCN Red List of Threatened Species 2014: e.T22918A16121857* [dataset]. International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.UK.2014-1.RLTS.T22918A16121857.en

Andriaholinirina, N., Baden, A., Blanco, M., Chikhi, L., Cooke, A., Davies, N., Dolch, R., Donati, G., Ganzhorn, J., Golden, C., Groeneveld, L. F., Rakotoarisoa, G., Rakotomanga, B., Rakotonanahary, J., Rakotondrainibe, H., Rakotondratsimba, G., Rakotondratsimba, M., Rakotonirina, L., Ralainasolo, F. B., … Zaramody, A. (2014). *Eulemur albifrons: The IUCN Red List of Threatened Species 2014: e.T8204A16116806* [dataset]. International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.UK.2014-1.RLTS.T8204A16116806.en

Angulo, E., Diagne, C., Ballesteros-Mejia, L., Adamjy, T., Ahmed, D. A., Akulov, E., Banerjee, A. K., Capinha, C., Dia, C. A. K. M., Dobigny, G., Duboscq-Carra, V. G., Golivets, M., Haubrock, P. J., Heringer, G., Kirichenko, N., Kourantidou, M., Liu, C., Nuñez, M. A., Renault, D., … Courchamp, F. (2021). Non-English languages enrich scientific knowledge: The example of economic costs of biological invasions. *Science of The Total Environment, 775*, 144441. https://doi.org/10.1016/j.scitotenv.2020.144441

Apanius, V., Yorinks, N., Bermingham, E., & Ricklefs, R. E. (2000). *Island and taxon effects in parasitism and resistance of Lesser Antillean birds.* http://repository.si.edu/xmlui/handle/10088/1255

Appenborn, R., Lammers, K., Harr, B., Lkhagvasuren, D., Samiya, R., Stubbe, M., & Ansorge, H. (2021). The house mouse *Mus musculus* in Mongolia: Taxonomy, status and ecology of a neglected species. *Erforschung Biologischer Ressourcen Der Mongolei / Exploration into the Biological Resources of Mongolia, ISSN 0440-1298*. https://digitalcommons.unl.edu/biolmongol/245

Appleton, C. C. (2003). Alien and invasive fresh water Gastropoda in South Africa. *African Journal of Aquatic Science, 28*(1), 69–81. https://doi.org/10.2989/16085914.2003.9626602

Appleton, C. C., & Brackenbury, T. D. (1998). Introduced freshwater gastropods in Africa with special reference to *Physa acuta*. In H. Madsen, C. C. Appleton, & M. Chimbari (Eds.), *Proceedings of a workshop on medical malacology in Africa* (pp. 22–26).

AquaNIS. (2015). *Information system on Aquatic Non-Indigenous and Cryptogenic Species.* World Wide Web Electronic Publication. Version 2.36+. Accessed 2021-02-19. www.corpi.ku.lt/databases/aquanis

Arenz, B. E., Held, B. W., Jurgens, J. A., & Blanchette, R. A. (2011). Fungal colonization of exotic substrates in Antarctica. *Fungal Diversity, 49*(1), 13–22. https://doi.org/10.1007/s13225-010-0079-4

Arianoutsou, M., Bazos, I., Delipetrou, P., & Kokkoris, Y. (2010). The alien flora of Greece: Taxonomy, life traits and habitat preferences. *Biological Invasions, 12*(10), Article 10. https://doi.org/10.1007/s10530-010-9749-0

Arianoutsou, M., Delipetrou, P., Vilà, M., Dimitrakopoulos, P. G., Celesti-Grapow, L., Wardell-Johnson, G., Henderson, L., Fuentes, N., Ugarte-Mendes, E., & Rundel, P. W. (2013). Comparative patterns of plant invasions in the Mediterranean biome. *PLoS ONE, 8*(11), e79174–e79174. https://doi.org/10.1371/journal.pone.0079174

Aristeidou, M., Herodotou, C., Ballard, H. L., Higgins, L., Johnson, R. F., Miller, A. E., Young, A. N., & Robinson, L. D. (2021). How do young community and citizen science volunteers support scientific research on biodiversity? The case of iNaturalist. *Diversity, 13*(7), 318. https://doi.org/10.3390/d13070318

Arlé, E., Zizka, A., Keil, P., Winter, M., Essl, F., Knight, T., Weigelt, P., Jiménez-Muñoz, M., & Meyer, C. (2021). bRacatus: A method to estimate the accuracy and biogeographical status of georeferenced biological data. *Methods in Ecology and Evolution, 12*(9), 1609–1619. https://doi.org/10.1111/2041-210X.13629

Arndt, E., Marchetti, M. P., & Schembri, P. J. (2018). Ecological impact of alien marine fishes: Insights from freshwater systems based on a comparative review. *Hydrobiologia, 817*(1), 457–474. https://doi.org/10.1007/s10750-018-3511-5

Arndt, E., & Schembri, P. J. (2015). Common traits associated with establishment and spread of Lessepsian fishes in the Mediterranean Sea. *Marine Biology, 162*(10), 2141–2153. https://doi.org/10.1007/s00227-015-2744-3

Aronson, M. F. J., Handel, S. N., La Puma, I. P., & Clemants, S. E. (2015). Urbanization promotes non-native woody species and diverse plant assemblages in the New York metropolitan region. *Urban Ecosystems, 18*(1), 31–45. https://doi.org/10.1007/s11252-014-0382-z

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M., Mörtberg, U., … Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences, 281*(1780), 20133330. https://doi.org/10.1098/rspb.2013.3330

Arteaga, M. A., Delgado, J. D., Otto, R., Fernández-Palacios, J. M., & Arévalo, J. R. (2009). How do alien plants distribute along roads on oceanic islands? A case study in Tenerife, Canary Islands. *Biological Invasions, 11*(4), 1071–1086. https://doi.org/10.1007/s10530-008-9329-8

Ascensão, F., Fahrig, L., Clevenger, A. P., Corlett, R. T., Jaeger, J. A. G., Laurance, W. F., & Pereira, H. M. (2018). Environmental challenges for the Belt and Road Initiative. *Nature Sustainability, 1*(5), 206–209. https://doi.org/10.1038/s41893-018-0059-3

Ascunce, M. S., Yang, C.-C., Oakey, J., Calcaterra, L., Wu, W.-J., Shih, C.-J., Goudet, J., Ross, K. G., & Shoemaker, D. (2011). Global Invasion History of the Fire Ant *Solenopsis invicta*. *Science, 331*(6020), 1066–1068. https://doi.org/10.1126/science.1198734

Ashton, I. W., Symstad, A. J., Davis, C. J., & Swanson, D. J. (2016). Preserving prairies: Understanding temporal and spatial patterns of invasive annual bromes in the Northern Great Plains. *Ecosphere, 7*(8). https://doi.org/10.1002/ecs2.1438

Atalah, J., & Sanchez-Jerez, P. (2020). Global assessment of ecological risks associated with farmed fish escapes. *Global Ecology and Conservation, 21*, e00842–e00842. https://doi.org/10.1016/j.gecco.2019.e00842

Atkinson, I. A. E. (1985). The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. In P. J. Moors (Ed.), *Conservation of Island Birds: Case studies for the management of threatened island species* (pp. 35–81). ICBP Technical Publication. http://www.hear.org/bibliography/references/49_atkinson_1985/

Atkinson, I. A. E. (1989). Introduced animals and extinctions. In D. Western & M. C. Pearl (Eds.), *Conservation for the twenty-first century.* Oxford University Press.

Audzijonyte, A., Wittmann, K. J., & Väinölä, R. (2007). Tracing recent invasions of the Ponto-Caspian mysid shrimp *Hemimysis anomala* across Europe and to North America with mitochondrial DNA. *Diversity and Distributions, 14*(2), 179–186. https://doi.org/10.1111/j.1472-4642.2007.00434.x

Aukema, J. E., McCullough, D. G., Holle, B. V., Liebhold, A. M., Britton, K., & Frankel, S. J. (2010). Historical accumulation of nonindigenous forest pests in the continental United States. *Bioscience, 60*(11), 886–897. https://doi.org/10.1525/bio.2010.60.11.5

Auld, B., Morita, H., Nishida, T., Ito, M., & Michael, P. (2003). Shared exotica: Plant invasions of Japan and south eastern Australia. *Cunninghamia, 8*(1), 147–152. https://scholar.google.com/scholar_lookup?hl=en&volume=8&publication_year=2003&pages=147-152&journal=%00null%00&issue=%00null%00&issn=%00null%00&author=B.+Auld&author=H.+Morita&author=T.+Nishida&author=M.+Ito&author=P.+Michael&title=Shared+exotica%3A+Plant+invasions+of+Japan+and+South+Eastern+Australia&pmid=%00empty%00&doi=%00empty%00

Auld, T. D., & Hutton, I. (2004). Conservation issues for the vascular flora of Lord Howe Island. *Cunninghamia : A Journal of Plant Ecology for Eastern Australia, 8*(4), 490–500. http://publikationen.ub.uni-frankfurt.de/frontdoor/index/index/docId/36883

Avila, G. A., Davidson, M., van Helden, M., & Fagan, L. (2019). The potential distribution of the Russian wheat aphid *(Diuraphis noxia)*: An updated distribution model including irrigation improves model fit for predicting potential spread. *Bulletin of Entomological Research, 109*(1), 90–101. https://doi.org/10.1017/S0007485318000226

Axmacher, J. C., & Sang, W. (2013). Plant Invasions in China—Challenges and Chances. *PLoS ONE, 8*(5). https://doi.org/10.1371/journal.pone.0064173

Azan, S., Bardecki, M., & Laursen, A. E. (2015). Invasive aquatic plants in the aquarium and ornamental pond industries: A risk assessment for southern O ntario (Canada). *Weed Research, 55*(3), 249–259. https://doi.org/10.1111/wre.12135

Bailey, S. A. (2015). An overview of thirty years of research on ballast water as a vector for aquatic invasive species to freshwater and marine environments. *Aquatic Ecosystem Health & Management, 18*(3), 261–268. https://doi.org/10.1080/14634988.2015.1027129

Bailey, S. A., Brown, L., Campbell, M. L., Canning-Clode, J., Carlton, J. T., Castro, N., Chainho, P., Chan, F. T., Creed, J. C., Curd, A., Darling, J., Fofonoff, P., Galil, B. S., Hewitt, C. L., Inglis, G. J., Keith, I., Mandrak, N. E., Marchini, A., McKenzie, C. H., … Zhan, A. (2020). Trends in the detection of aquatic non-indigenous species across global marine, estuarine and freshwater ecosystems: A 50-year perspective. *Diversity and Distributions, 26*(12), 1780–1797. https://doi.org/10.1111/ddi.13167

Baird, H. P., Janion-Scheepers, C., Stevens, M. I., Leihy, R. I., & Chown, S. L. (2019). The ecological biogeography of indigenous and introduced Antarctic springtails. *Journal of Biogeography, 46*(9), 1959–1973. https://doi.org/10.1111/jbi.13639

Ballard, H. L., Dixon, C. G. H., & Harris, E. M. (2017). Youth-focused citizen science: Examining the role of environmental science learning and agency for conservation. *Biological Conservation, 208*, 65–75. https://doi.org/10.1016/j.biocon.2016.05.024

Ballard, H. L., Robinson, L. D., Young, A. N., Pauly, G. B., Higgins, L. M., Johnson, R. F., & Tweddle, J. C. (2017). Contributions to conservation outcomes by natural history museum-led citizen science: Examining evidence and next steps. *Biological Conservation, 208*, 87–97. https://doi.org/10.1016/j.biocon.2016.08.040

Balon, E. (1995). Origin and domestication of the wild carp, *Cyprinus carpio*: From Roman gourmets to the swimming flowers. *Aquaculture, 129*, 3–48. https://doi.org/10.1016/0044-8486(94)00227-F

Banerjee, A. K. (2020). *ILORA_datasets* [dataset]. Harvard Dataverse. https://doi.org/10.7910/DVN/VUPNIT

Banerjee, A. K., Khuroob, A. A., Dehnen-Schmutz, K., Pant, V., Patwardhan, C., Bhowmick, A. R., & Mukherjee, A. (2021). An integrated policy framework and plan of action to prevent and control plant invasions in India. *Environmental Science and Policy, 124*, 64–72. https://doi.org/10.1016/j.envsci.2021.06.003

Bang, A., & Courchamp, F. (2021). Industrial rearing of edible insects could be a major source of new biological invasions. *Ecology*

Letters, 24(3), 393–397. https://doi.org/10.1111/ele.13646

Baret, S., Rouget, M., Richardson, D. M., Lavergne, C., Egoh, B., Dupont, J., & Strasberg, D. (2006). Current distribution and potential extent of the most invasive alien plant species on La Réunion (Indian Ocean, Mascarene islands). *Austral Ecology, 31*(6), 747–758. https://doi.org/10.1111/j.1442-9993.2006.01636.x

Bariche, M., & Fricke, R. (2020). The marine ichthyofauna of Lebanon: An annotated checklist, history, biogeography, and conservation status. *Zootaxa, 4775*(1), 1–157. https://doi.org/10.11646/zootaxa.4775.1.1

Barina, Z., Rakaj, M., Somogyi, G., Erős-Honti, Z., & Pifkó, D. (2014). The alien flora of Albania: History, current status and future trends. *Weed Research, 54*(2), 196–215. https://doi.org/10.1111/wre.12061

Barker, G. M. (1999). Naturalised terrestrial Stylommatophora (Mollusca: Gastropoda). *Fauna of New Zealand, 38*, 1–253. https://doi.org/10.7931/J2/FNZ.38

Bartlett, J. C., Convey, P., & Hayward, S. A. L. (2020). Surviving the Antarctic Winter—Life Stage Cold Tolerance and Ice Entrapment Survival in The Invasive Chironomid Midge *Eretmoptera murphyi. Insects, 11*(3), 147. https://doi.org/10.3390/insects11030147

Bartley, D. M., & Marttin, F. (2004). Introduction of Alien Species/Strains and Their Impact on Biodiversity. In M. V. Gupta, D. M. Bartley, & B. O. Acosta (Eds.), *Use of Genetically Improved and Alien Species for Aquaculture and Conservation of Aquatic Biodiversity in Africa* (pp. 16–21). https://portals.iucn.org/library/sites/library/files/documents/2004-048.pdf

Bartomeus, I., Ascher, J. S., Gibbs, J., Danforth, B. N., Wagner, D. L., Hedtke, S. M., & Winfree, R. (2013). Historical changes in northeastern US bee pollinators related to shared ecological traits. *Proceedings of the National Academy of Sciences, 110*(12), 4656–4660. https://doi.org/10.1073/pnas.1218503110

Barwell, L. J., Perez-Sierra, A., Henricot, B., Harris, A., Burgess, T. I., Hardy, G., Scott, P., Williams, N., Cooke, D. E. L., Green, S., Chapman, D. S., & Purse, B. V. (2021). Evolutionary trait-based approaches for predicting future global impacts of plant pathogens in the genus *Phytophthora. Journal of Applied Ecology, 58*(4), Article n/a. https://doi.org/10.1111/1365-2664.13820

Bebber, D. P. (2015). Range-Expanding Pests and Pathogens in a Warming World. *Annual Review of Phytopathology, 53*(1), Article 1. https://doi.org/10.1146/annurev-phyto-080614-120207

Bebber, D. P., Field, E., Gui, H., Mortimer, P., Holmes, T., & Gurr, S. J. (2019). Many unreported crop pests and pathogens are probably already present. *Global Change Biology, 25*(8), 2703–2713. https://doi.org/10.1111/gcb.14698

Bebber, D. P., Holmes, T., Smith, D., & Gurr, S. J. (2014). Economic and physical determinants of the global distributions of crop pests and pathogens. *New Phytologist, 202*(3), 901–910. https://doi.org/10.1111/nph.12722

Bebber, D. P., Ramotowski, M. A. T., & Gurr, S. J. (2013). Crop pests and pathogens move polewards in a warming world. *Nature Climate Change, 3*(11), Article 11. https://doi.org/10.1038/nclimate1990

Becker, T., Dietz, H., Billeter, R., Buschmann, H., & Edwards, P. J. (2005). Altitudinal distribution of alien plant species in the Swiss Alps. *Perspectives in Plant Ecology, Evolution and Systematics, 7*(3), Article 3. https://doi.org/10.1016/j.ppees.2005.09.006

Beggs, J. R., Brockerhoff, E. G., Corley, J. C., Kenis, M., Masciocchi, M., Muller, F., Rome, Q., & Villemant, C. (2011). Ecological effects and management of invasive alien Vespidae. *BioControl, 56*(4), 505–526. https://doi.org/10.1007/s10526-011-9389-z

Bekele, K., Haji, J., Legesse, B., & Schaffner, U. (2018). Economic impacts of *Prosopis* spp. invasions on dryland ecosystem services in Ethiopia and Kenya: Evidence from choice experimental data. *Journal of Arid Environments, 158*, 9–18. https://doi.org/10.1016/j.jaridenv.2018.07.001

Bellard, C., Genovesi, P., & Jeschke, J. M. (2016). Global patterns in threats to vertebrates by biological invasions. *Proceedings of the Royal Society B: Biological Sciences, 283*(1823), 20152454. https://doi.org/10.1098/rspb.2015.2454

Bellard, C., & Jeschke, J. M. (2016). A spatial mismatch between invader impacts and research publications. *Conservation Biology, 30*(1), 230–232. https://doi.org/10.1111/cobi.12611

Bellard, C., Leclerc, C., & Courchamp, F. (2013). Potential impact of sea level rise on French islands worldwide. *Nature Conservation, 5*, 75–86. https://doi.org/10.3897/natureconservation.5.5533

Bellard, C., Leclerc, C., & Courchamp, F. (2014). Impact of sea level rise on the 10 insular biodiversity hotspots. *Global Ecology and Biogeography, 23*(2), 203–212. https://doi.org/10.1111/geb.12093

Bellard, C., Rysman, J.-F., Leroy, B., Claud, C., & Mace, G. M. (2017). A global picture of biological invasion threat on islands. *Nature Ecology & Evolution, 1*(12), 1862–1869. https://doi.org/10.1038/s41559-017-0365-6

Bellard, C., Thuiller, W., Leroy, B., Genovesi, P., Bakkenes, M., & Courchamp, F. (2013). Will climate change promote future invasions? *Global Change Biology, 19*(12), 3740–3748. https://doi.org/10.1111/gcb.12344

Belmaker, J., Brokovich, E., China, V., Golani, D., & Kiflawi, M. (2009). Estimating the rate of biological introductions: Lessepsian fishes in the Mediterranean. *Ecology, 90*(4), 1134–1141. https://doi.org/10.1890/07-1904.1

Bender, N. A., Crosbie, K., & Lynch, H. J. (2016). Patterns of tourism in the Antarctic Peninsula region: A 20-year analysis. *Antarctic Science, 28*(3), 194–203. https://doi.org/10.1017/S0954102016000031

Bennett, J. R., Shaw, J. D., Terauds, A., Smol, J. P., Aerts, R., Bergstrom, D. M., Blais, J. M., Cheung, W. W., Chown, S. L., Lea, M.-A., Nielsen, U. N., Pauly, D., Reimer, K. J., Riddle, M. J., Snape, I., Stark, J. S., Tulloch, V. J., & Possingham, H. P. (2015). Polar lessons learned: Long-term management based on shared threats in Arctic and Antarctic environments. *Frontiers in Ecology and the Environment, 13*(6), 316–324. https://doi.org/10.1890/140315

Berger, L., Roberts, A. A., Voyles, J., Longcore, J. E., Murray, K. A., & Skerratt, L. F. (2016). History and recent progress on chytridiomycosis in amphibians. *Fungal Ecology, 19*, 89–99. https://doi.org/10.1016/j.funeco.2015.09.007

Bernery, C., Bellard, C., Courchamp, F., Brosse, S., Gozlan, R. E., Jarić, I., Teletchea, F., & Leroy, B. (2022). Freshwater fish invasions: A comprehensive review. *Annual Review of Ecology, Evolution, and Systematics, 53*, 427–456. https://doi.org/10.1146/annurev-ecolsys-032522-015551

Bertelsmeier, C., Blight, O., & Courchamp, F. (2016). Invasions of ants (Hymenoptera: Formicidae) in light of global climate change. *Myrmecological News, 22*, 25–42. https://www.semanticscholar.org/paper/Invasions-of-ants-(Hymenoptera%3A-Formicidae)-in-of-Bertelsmeier-Blight/c77c939272321ad7cfec4c0d80f12544f427b3e8

Bertelsmeier, C., Luque, G. M., & Courchamp, F. (2013a). Increase in Quantity and Quality of Suitable Areas for Invasive Species as Climate Changes. *Conservation Biology, 27*(6), 1458–1467. https://doi.org/10.1111/cobi.12093

Bertelsmeier, C., Luque, G. M., & Courchamp, F. (2013b). The Impact of Climate Change Changes over Time. *Biological Conservation, 167*, 107–115. https://doi.org/10.1016/j.biocon.2013.07.038

Bertelsmeier, C., Luque, G. M., Hoffmann, B. D., & Courchamp, F. (2015). Worldwide ant invasions under climate change. *Biodiversity and Conservation, 24*(1), 117–128. https://doi.org/10.1007/s10531-014-0794-3

Bertelsmeier, C., Ollier, S., Liebhold, A., & Keller, L. (2017). Recent human history governs global ant invasion dynamics. *Nature Ecology & Evolution, 1*(7), Article 7. https://doi.org/10.1038/s41559-017-0184

Bertelsmeier, C., Ollier, S., Liebhold, A. M., Brockerhoff, E. G., Ward, D., & Keller, L. (2018). Recurrent bridgehead effects accelerate global alien ant spread. *Proceedings of the National Academy of Sciences, 115*(21), 5486–5491. https://doi.org/10.1073/pnas.1801990115

Bertness, M. D. (1984). Habitat and community modification by an introduced herbivorous snail. *Ecology, 65*(2), 370–381. https://doi.org/10.2307/1941400

Bertolino, S., di Montezemolo, N. C., Preatoni, D. G., Wauters, L. A., & Martinoli, A. (2014). A grey future for Europe: *Sciurus carolinensis* is replacing native red squirrels in Italy. *Biological Invasions, 16*(1), 53–62. https://doi.org/10.1007/s10530-013-0502-3

Bertolino, S., Lurz, P. W. W., Sanderson, R., & Rushton, S. P. (2008). Predicting the spread of the American grey squirrel *(Sciurus carolinensis)* in Europe: A call for a co-ordinated European approach. *Biological Conservation, 141*(10), 2564–2575. https://doi.org/10.1016/j.biocon.2008.07.017

Berzitis, E. A., Minigan, J. N., Hallett, R. H., & Newman, J. A. (2014). Climate and host plant availability impact the future distribution of the bean leaf beetle *(Cerotoma trifurcata)*. *Global Change Biology, 20*(9), Article 9. https://doi.org/10.1111/gcb.12557

Bezerra, L. A. V., Freitas, M. O., Daga, V. S., Occhi, T. V. T., Faria, L., Costa, A. P. L., Padial, A. A., Prodocimo, V., & Vitule, J. R. S. (2019). A network meta-analysis of threats to South American fish biodiversity. *Fish and Fisheries, 20*(4), 620–639. https://doi.org/10.1111/faf.12365

Bhatta, S., Joshi, L. R., & Shrestha, B. B. (2020). Distribution and impact of invasive alien plant species in Bardia National Park, western Nepal. *Environmental Conservation, 47*(3), 197–205. https://doi.org/10.1017/S0376892920000223

Biancolini, D., Vascellari, V., Melone, B., Blackburn, T. M., Cassey, P., Scrivens, S. L., & Rondinini, C. (2021). DAMA: the global Distribution of Alien Mammals database. *Ecology, 102*(11). https://doi.org/10.1002/ecy.3474

Bilecenoğlu, M., & Çınar, M. E. (2021). Alien Species Threat across Marine Protected Areas of Turkey—An Updated Inventory. *Journal of Marine Science and Engineering, 9*(10), Article 10. https://doi.org/10.3390/jmse9101077

Blackburn, T. M., Cassey, P., & Lockwood, Julie. L. (2008). The island biogeography of exotic bird species. *Global Ecology and Biogeography, 17*(2), 246–251. https://doi.org/10.1111/j.1466-8238.2007.00361.x

Blackburn, T. M., Delean, S., Pyšek, P., & Cassey, P. (2016). On the island biogeography of aliens: A global analysis of the richness of plant and bird species on oceanic islands: The island biogeography of alien species. *Global Ecology and Biogeography, 25*(7), 859–868. https://doi.org/10.1111/geb.12339

Blackburn, T. M., Dyer, E. E., Su, S., & Cassey, P. (2015). Long after the event, or four things we (should) know about bird invasions. *Journal of Ornithology, 156*(1), 15–25. https://doi.org/10.1007/s10336-015-1155-z

Blackburn, T. M., Scrivens, S. L., Heinrich, S., & Cassey, P. (2017). Patterns of selectivity in introductions of mammal species worldwide. *NeoBiota, 33*(1), 33–51. https://doi.org/10.3897/neobiota.33.10471

Blair, R. B., & Johnson, E. M. (2008). Suburban habitats and their role for birds in the urban–rural habitat network: Points of local invasion and extinction? *Landscape Ecology, 23*(10), 1157–1169. https://doi.org/10.1007/s10980-008-9267-y

Blakeslee, A. M. H., Kamakura, Y., Onufrey, J., Makino, W., Urabe, J., Park, S., Keogh, C. L., Miller, A. W., Minton, M. S., Carlton, J. T., & Miura, O. (2017). Reconstructing the Invasion History of the Asian shorecrab, *Hemigrapsus sanguineus* (De Haan 1835) in the Western Atlantic. *Marine Biology, 164*(3), 47. https://doi.org/10.1007/s00227-017-3069-1

Blanchard, M. (2009). Recent expansion of the slipper limpet population *(Crepidula fornicata)* in the Bay of Mont-Saint-Michel (Western Channel, France). *Aquatic Living Resources, 22*(1), 11–19. https://doi.org/10.1051/alr/2009004

Blanchet, S., Grenouillet, G., Beauchard, O., Tedesco, P. A., Leprieur, F., Dürr, H. H., Busson, F., Oberdorff, T., & Brosse, S. (2010). Non-native species disrupt the worldwide patterns of freshwater fish body size: Implications for Bergmann's rule. *Ecology Letters, 13*(4), 421–431. https://doi.org/10.1111/j.1461-0248.2009.01432.x

Blanco, A., Neto, J. M., Trancoso, J., Lemos, M. F. L., & Olabarria, C. (2020). Effectiveness of two western Iberian Peninsula marine protected areas in reducing the risk of macroalgae invasion. *Ecological Indicators, 108*. https://doi.org/10.1016/j.ecolind.2019.105705

Blehert, D. S., Hicks, A. C., Behr, M., Meteyer, C. U., Berlowski-Zier, B. M., Buckles, E. L., Coleman, J. T. H., Darling, S. R., Gargas, A., Niver, R., Okoniewski, J. C., Rudd, R. J., & Stone, W. B. (2009). Bat white-nose syndrome: An emerging fungal pathogen? *Science, 323*(5911), 227. https://doi.org/10.1126/science.1163874

Bock, D. G., Caseys, C., Cousens, R. D., Hahn, M. A., Heredia, S. M., Hübner, S., Turner, K. G., Whitney, K. D., & Rieseberg, L. H. (2015). What we still don't know about invasion genetics. *Molecular Ecology, 24*(9), 2277–2297. https://doi.org/doi:10.1111/mec.13032

Boehmer, H. J. (2011). Vulnerability of tropical montane rain forest ecosystems due to climate change. In H. G. Brauch, Ú. Oswald Spring, C. Mesjasz, J. Grin, P. Kameri-Mbote, B. Chourou, P. Dunay, & J. Birkmann (Eds.), *Coping with Global Environmental Change, Disasters and Security: Threats, Challenges, Vulnerabilities and Risks* (Vol. 5). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-17776-7

Boersma, M., Malzahn, A. M., Greve, W., & Javidpour, J. (2007). The first occurrence of the ctenophore *Mnemiopsis leidyi* in the North Sea. *Helgoland Marine Research, 61*(2), Article 2. https://doi.org/10.1007/s10152-006-0055-2

Böhm, M., Collen, B., Baillie, J. E. M., Bowles, P., Chanson, J., Cox, N., Hammerson, G., Hoffmann, M., Livingstone, S. R., Ram, M., Rhodin, A. G. J., Stuart, S. N., van Dijk, P. P., Young, B. E., Afuang, L. E., Aghasyan, A., García, A., Aguilar, C., Ajtic, R., … Zug, G. (2013). The conservation status of the world's reptiles. *Biological Conservation, 157*, 372–385. https://doi.org/10.1016/j.biocon.2012.07.015

Boltovskoy, D., Correa, N., Cataldo, D., & Sylvester, F. (2006). Dispersion and Ecological Impact of the Invasive Freshwater Bivalve *Limnoperna fortunei* in the Río de la Plata Watershed and Beyond. *Biological Invasions, 8*(4), 947–963. https://doi.org/10.1007/s10530-005-5107-z

Bomford, M., Kraus, F., Barry, S. C., & Lawrence, E. (2009). Predicting establishment success for alien reptiles and amphibians: A role for climate matching. *Biological Invasions, 11*(3), 713–724. https://doi.org/10.1007/s10530-008-9285-3

Bonesi, L., & Palazon, S. (2007). The American mink in Europe: Status, impacts, and control. *Biological Conservation, 134*(4), 470–483. https://doi.org/10.1016/j.biocon.2006.09.006

Bonnamour, A., Gippet, J. M. W., & Bertelsmeier, C. (2021). Insect and plant invasions follow two waves of globalisation. *Ecology Letters, 24*(11), 2418–2426. https://doi.org/10.1111/ele.13863

Bonnaud, E., Medina, F. M., Vidal, E., Nogales, M., Tershy, B., Zavaleta, E., Donlan, C. J., Keitt, B., Le Corre, M., & Horwath, S. V. (2011). The diet of feral cats on islands: A review and a call for more studies. *Biological Invasions, 13*(3), 581–603. https://doi.org/10.1007/s10530-010-9851-3

Boo, G. H., Mansilla, A., Nelson, W., Bellgrove, A., & Boo, S. M. (2014). Genetic connectivity between trans-oceanic populations of *Capreolia implexa* (Gelidiales, Rhodophyta) in cool temperate waters of Australasia and Chile. *Aquatic Botany, 119*, 73–79. https://doi.org/10.1016/j.aquabot.2014.08.004

Booth, B. D., Murphy, S. D., & Swanton, C. J. (Eds.). (2003). *Weed ecology in natural and agricultural systems* (1st ed.). CABI Publishing. https://doi.org/10.1079/9780851995281.0000

Borroto-Páez, R., Bosch, R. A., Fabres, B. A., & García, O. A. (2015). Introduced amphibians and reptiles in the Cuban archipelago. *Herpetological Conservation and Biology, 10*(3), 985–1012. https://www.herpconbio.org/Volume_10/Issue_3/Borroto-Paez_etal_2015.pdf

Bortolus, A., Adam, P., Adams, J. B., Ainouche, M. L., Ayres, D., Bertness, M. D., Bouma, T. J., Bruno, J. F., Caçador, I., Carlton, J. T., Castillo, J. M., Costa, C. S. B., Davy, A. J., Deegan, L., Duarte, B., Figueroa, E., Gerwein, J., Gray, A. J., Grosholz, E. D., … Weis, J. S. (2019). Supporting *Spartina*: Interdisciplinary perspective shows *Spartina* as a distinct solid genus. *Ecology, 100*(11), e02863. https://doi.org/10.1002/ecy.2863

Bortolus, A., Carlton, J. T., & Schwindt, E. (2015). Reimagining South American coasts: Unveiling the hidden invasion history of an iconic ecological engineer. *Diversity and Distributions, 21*(11), 1267–1283. https://doi.org/10.1111/ddi.12377

Bortolus, A., & Schwindt, E. (2007). What would Darwin have written now? *Biodiversity and Conservation, 16*(2), 337–345. https://doi.org/10.1007/s10531-005-1874-1

Bossdorf, O., Auge, H., Lafuma, L., Rogers, W. E., Siemann, E., & Prati, D. (2005). Phenotypic and genetic differentiation between native and introduced plant populations. *Oecologia, 144*(1), 1–11. https://doi.org/10.1007/s00442-005-0070-z

Bossdorf, O., Richards, C. L., & Pigliucci, M. (2008). Epigenetics for ecologists. *Ecology Letters, 11*(2), 106–115. https://doi.org/10.1111/j.1461-0248.2007.01130.x

Bourdôt, G. W., Lamoureaux, S. L., Watt, M. S., Manning, L. K., & Kriticos, D. J. (2012). The potential global distribution of the invasive weed Nassella neesiana under current and future climates. *Biological Invasions, 14*(8), 1545–1556. https://doi.org/10.1007/s10530-010-9905-6

Bourlière, F., & Hadley, M. (1983). Present-day savannas: An overview. In F. Bourlière (Ed.), *Ecosystems of the world 13. Tropical savannas.* Elsevier. https://eunis.eea.europa.eu/references/69

Bowen, J. L., Kearns, P. J., Byrnes, J. E. K., Wigginton, S., Allen, W. J., Greenwood, M., Tran, K., Yu, J., Cronin, J. T., & Meyerson, A. (2017). Lineage overwhelms environmental conditions in determining rhizosphere bacterial community structure in a cosmopolitan invasive plant. *Nature Communications, 8*(433). https://doi.org/10.1038/s41467-017-00626-0

Bradley, M. J., Kutz, S. J., Jenkins, E., & O'Hara, T. M. (2005). The potential impact of climate change on infectious diseases of Arctic fauna. *International Journal of Circumpolar Health, 64*(5), 468–477. https://doi.org/10.3402/ijch.v64i5.18028

Branch, G. M., & Steffani, C. N. (2004). Can we predict the effects of alien species? A case-history of the invasion of South Africa by *Mytilus galloprovincialis* (Lamarck). *Journal of Experimental Marine Biology and Ecology, 300*(1–2), 189–215. https://doi.org/10.1016/j.jembe.2003.12.007

Branco, M., Nunes, P., Roques, A., Fernandes, M. R., Orazio, C., & Jactel, H. (2019). Urban trees facilitate the establishment of non-native forest insects. *NeoBiota, 52*, 25–46. https://doi.org/10.3897/neobiota.52.36358

Brandeis, T. J., Helmer, E. H., Marcano-Vega, H., & Lugo, A. E. (2009). Climate shapes the novel plant communities that form after deforestation in Puerto Rico and the U.S. Virgin Islands. *Forest Ecology and Management, 258*(7), 1704–1718. https://doi.org/10.1016/j.foreco.2009.07.030

Brasier, C., & Kirk, S. (2000). Survival of clones of NAN *Ophiostoma novo-ulmi* around its probable centre of appearance in North America. *Mycological Research, 104*(11), 1322–1332. https://doi.org/10.1017/S0953756200002732

Braun, M., Schindler, S., & Essl, F. (2016). Distribution and management of invasive alien plant species in protected areas in Central Europe. *Journal for Nature Conservation, 33*, 48–57. https://doi.org/10.1016/j.jnc.2016.07.002

Brenton-Rule, E. C., Barbieri, R. F., & Lester, P. J. (2016). Corruption, development and governance indicators predict invasive species risk from trade. *Proceedings of the Royal Society B: Biological Sciences, 283*(1832), 20160901. https://doi.org/10.1098/rspb.2016.0901

Briggs, M., Funge-Smith, S., Subasinghe, R., & Phillips, M. (2004). 6. Threats and risks of introducing alien shrimp species. In *Introductions and movement of Penaeus vannamei and Penaeus stylirostris in Asia and the Pacific.* Food and Agriculture Organization of The United Nations Regional Office for Asia And the Pacific. https://www.fao.org/3/ad505e/ad505e08.htm

Brodie, G. D., Barker, G. M., Stevens, F., & Fiu, M. (2014). Preliminary re-survey of the land snail fauna of Rotuma: Conservation and biosecurity implications. *Pacific Conservation Biology, 20*(1), Article 1. https://doi.org/10.1071/PC140094

Brook, R. K., & van Beest, F. M. (2014). Feral wild boar distribution and perceptions of risk on the central Canadian prairies: Distribution and Risk Perceptions of Prairie Boar. *Wildlife Society Bulletin, 38*(3), 486–494. https://doi.org/10.1002/wsb.424

Brooks, E. G. E., Allen, D. J., & Darwall, W. R. T. (2011). *The Status and Distribution of Freshwater Biodiversity in Central Africa.* IUCN. https://portals.iucn.org/library/node/9825

Brotz, L., Cheung, W. W. L., Kleisner, K., Pakhomov, E., & Pauly, D. (2012). Increasing jellyfish populations: Trends in Large Marine Ecosystems. *Hydrobiologia, 690*, 3–20. https://doi.org/10.1007/s10750-012-1039-7

Brown, C. J., Blossey, B., Maerz, J. C., & Joule, S. J. (2006). Invasive plant and experimental venue affect tadpole performance. *Biological Invasions, 8*(2), 327–338. https://doi.org/10.1007/s10530-004-8244-x

Brown, J. K. M., & Hovmøller, M. S. (2002). Aerial dispersal of pathogens on the global and continental scales and its impact on plant disease. *Science, 297*(5581), 537–541. https://doi.org/10.1126/science.1072678

Brundu, G. (2015a). Council of Europe's Code of Conducts for preventing and mitigating impact of invasive alien plants. *Italian Botanist, 47*(2), 356–360. https://www.scopus.com/inward/record.uri?eid=2-s2.0-85050010167&partnerID=40&md5=0a63b64e29e6340d8798556e4a659553

Brundu, G. (2015b). Plant invaders in European and Mediterranean inland waters: Profiles, distribution, and threats. *Hydrobiologia, 746*(1), 61–79. https://doi.org/10.1007/s10750-014-1910-9

Brundu, G., & Camarda, I. (2013). The Flora of Chad: A checklist and brief analysis. *PhytoKeys, 23*, 1–18. https://doi.org/10.3897/phytokeys.23.4752

Bryant, J. E., Holmes, E. C., & Barrett, A. D. T. (2007). Out of Africa: A molecular perspective on the introduction of yellow fever virus into the Americas. *PLoS Pathogens, 3*(5), e75. https://doi.org/10.1371/journal.ppat.0030075

Bucher, E. H., & Aramburú, R. M. (2014). Land-use changes and monk parakeet expansion in the Pampas grasslands of Argentina. *Journal of Biogeography, 41*(6), 1160–1170. https://doi.org/10.1111/jbi.12282

Buczkowski, G., & Bertelsmeier, C. (2017). Invasive termites in a changing climate: A global perspective. *Ecology and Evolution, 7*(3), 974–985. https://doi.org/10.1002/ece3.2674

Burgess, T. I., Scott, J. K., Mcdougall, K. L., Stukely, M. J. C., Crane, C., Dunstan, W. A., Brigg, F., Andjic, V., White, D., Rudman, T., Arentz, F., Ota, N., & Hardy, G. E. St. J. (2017). Current and projected global distribution of *Phytophthora cinnamomi*, one of the world's worst plant pathogens. *Global Change Biology, 23*(4), 1661–1674. https://doi.org/10.1111/gcb.13492

Bustamante, D. E., & Ramírez, M. E. (2009). El género *Polysiphonia sensu lato*, en la costa norte y centro-sur de Chile (18-41os) (Rhodophyta, Rhodomelaceae). *Boletín del Museo Nacional de Historia Natural, Chile, 58*, 31–50. https://publicaciones.mnhn.gob.cl/668/articles-64566_archivo_01.pdf

Bustamante, R. O., & Simonetti, J. A. (2005). Is *Pinus radiata* invading the native vegetation in central Chile? Demographic responses in a fragmented forest. *Biological Invasions, 7*(2), 243–249. https://doi.org/10.1007/s10530-004-0740-5

Byers, J. E. (2005). Marine Reserves Enhance Abundance but Not Competitive Impacts of a Harvested Nonindigenous Species. *Ecology, 86*(2), 487–500. https://doi.org/10.1890/03-0580

Cacabelos, E., Martins, G. M., Faria, J., Prestes, A. C. L., Costa, T., Moreu, I., & Neto, A. I. (2020). Limited effects of marine protected areas on the distribution of invasive species, despite positive effects on diversity in shallow-water marine communities. *Biological Invasions, 22*(3), 1169–1179. https://doi.org/10.1007/s10530-019-02171-x

Cadien, D. B., & Ranasinghe, J. A. (2003). Invaders in the open sea: Establishment of the New Zealand snail *Philine auriformis* in southern California coastal waters. *Environmental Science*. https://www.semanticscholar.org/paper/Invaders-in-the-open-sea-%3A-Establishment-of-the-New-Cadien-Ranasinghe/4b50931639b2b9afa38e1b6382e854ee6f497463

Cadotte, M. W. (2021). The list of vascular plants for the city of Toronto. *Ecological Solutions and Evidence, 2*(1), e12036. https://doi.org/10.1002/2688-8319.12036

Cadotte, M. W., Yasui, S. L. E., Livingstone, S., & MacIvor, J. S. (2017). Are urban systems beneficial, detrimental, or indifferent for biological invasion? *Biological Invasions, 19*(12), 3489–3503. https://doi.org/10.1007/s10530-017-1586-y

Calder, D. R., Carlton, J. T., Keith, I., Larson, K., McCann, L., Geller, J., Wheelock, M., Choong, H. H. C., & Ruiz, G. M. (2021). Additions to the hydroids (Cnidaria, Hydrozoa) of marine fouling communities on the mainland of Ecuador and in the Galapagos Islands. *Aquatic Invasions, 16*(2), 208–252. https://doi.org/10.3391/ai.2021.16.2.02

Campagnaro, T., Brundu, G., & Sitzia, T. (2018). Five major invasive alien tree species in European Union forest habitat types of the Alpine and Continental biogeographical regions. *Journal for Nature Conservation, 43*, 227–238. https://doi.org/10.1016/j.jnc.2017.07.007

Campbell, D. J., & Atkinson, I. A. E. (2002). Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands. *Biological Conservation, 107*(1), 19–35. https://doi.org/10.1016/S0006-3207(02)00039-3

Campbell, F. (2010). *Erythrina Gall Wasp*: Quadrastichus erythrinae. Don't Move Firewood. https://www.dontmovefirewood.org/pest_pathogen/erythrina-gall-wasp-html/

Campbell, M. L., Hewitt, C. L., & Miles, J. (2016). Marine pests in paradise: Capacity building, awareness raising and preliminary introduced species port survey results in the Republic of Palau. *Management of Biological Invasions, 7*(4), 351–363. https://doi.org/10.3391/mbi.2016.7.4.05

Campbell, M. L., King, S., Heppenstall, L. D., van Gool, E., Martin, R., & Hewitt, C. L. (2017). Aquaculture and urban marine structures facilitate native and non-indigenous species transfer through generation and accumulation of marine debris. *Marine Pollution Bulletin, 123*(1–2), 304–312. https://doi.org/10.1016/j.marpolbul.2017.08.040

Canavan, S., Meyerson, L. A., Packer, J. G., Pyšek, P., Maurel, N., Lozano, V., Richardson, D. M., Brundu, G., Canavan, K., Cicatelli, A., Čuda, J., Dawson, W., Essl, F., Guarino, F., Guo, W.-Y., van Kleunen, M., Kreft, H., Lambertini, C., Pergl, J., … Wilson, J. R. U. (2019). Tall-statured grasses: A useful functional group for invasion science. *Biological Invasions, 21*(1), 37–58. https://doi.org/10.1007/s10530-018-1815-z

Canning-Clode, J., & Carlton, J. T. (2017). Refining and expanding global climate change scenarios in the sea: Poleward creep complexities, range termini, and setbacks and surges. *Diversity and Distributions, 23*(5), 463–473. https://doi.org/10.1111/ddi.12551

Canning-Clode, J., Fowler, A. E., Byers, J. E., Carlton, J. T., & Ruiz, G. M. (2011). *'Caribbean Creep'* Chills Out: Climate Change and Marine Invasive Species. *PLoS ONE, 6*(12), e29657. https://doi.org/10.1371/journal.pone.0029657

Canonico, G. C., Arthington, A., McCrary, J. K., & Thieme, M. L. (2005). The effects of introduced tilapias on native biodiversity. *Aquatic Conservation: Marine and Freshwater Ecosystems, 15*(5), 463–483. https://doi.org/10.1002/aqc.699

Cao, L., Naylor, R., Henriksson, P., Leadbitter, D., Metian, M., Troell, M., & Zhang, W. (2015). China's aquaculture and the world's wild fisheries. *Science, 347*(6218), 133–135. https://doi.org/10.1126/science.1260149

Capinha, C., Essl, F., Seebens, H., Moser, D., & Pereira, H. M. (2015). The dispersal of alien species redefines biogeography in the Anthropocene. *Science, 348*(6240), 1248–1251. https://doi.org/10.1126/science.aaa8913

Capinha, C., Rocha, J., & Sousa, C. A. (2014). Macroclimate Determines the Global Range Limit of *Aedes aegypti*. *EcoHealth, 11*(3), 420–428. https://doi.org/10.1007/s10393-014-0918-y

Capinha, C., Seebens, H., Cassey, P., García-Díaz, P., Lenzner, B., Mang, T., Moser, D., Pyšek, P., Rödder, D., Scalera, R., Winter, M., Dullinger, S., & Essl, F. (2017). Diversity, biogeography and the global flows of alien amphibians and reptiles. *Diversity and Distributions, 23*(11), 1313–1322. https://doi.org/10.1111/ddi.12617

Capizzi, D. (2020). A review of mammal eradications on Mediterranean islands. *Mammal Review, 50*(2), 124–135. https://doi.org/10.1111/mam.12190

Carboni, M., Guéguen, M., Barros, C., Georges, D., Boulangeat, I., Douzet, R., Dullinger, S., Klonner, G., Kleunen, M. van, Essl, F., Bossdorf, O., Haeuser, E., Talluto, M. V., Moser, D., Block, S., Conti, L., Dullinger, I., Münkemüller, T., & Thuiller, W. (2018). Simulating plant invasion dynamics in mountain ecosystems under global change scenarios. *Global Change Biology, 24*(1), Article 1. https://doi.org/10.1111/gcb.13879

Cárdenas, L., Leclerc, J.-C., Bruning, P., Garrido, I., Détrée, C., Figueroa, A., Astorga, M., Navarro, J. M., Johnson, L. E., Carlton, J. T., & Pardo, L. (2020). First mussel settlement observed in Antarctica reveals the potential for future invasions. *Scientific Reports, 10*(1), 5552. https://doi.org/10.1038/s41598-020-62340-0

Cárdenas-Calle, M., Perez-Correa, J., Martinez, P., Keith, I., Rivera, F., Cornejo, M., Torres, G. M., Villamar, F., Zambrano, R., Cárdenas, A., Triviño, M., Troccoli, L., Bigatti, G., Coronel, J., & Mora, E. (2019). First report of marine alien species in mainland Ecuador: Threats of invasion in rocky shores. In C. R. Veitch, M. N. Clout, A. R. Martin, J. C. Russell, & C. J. West (Eds.), *Island invasives: Scaling up to meet the challenge* (pp. 452–457). IUCN. https://doi.org/10.2305/IUCN.CH.2019.SSC-OP.62.en

Carey, M. P., Sethi, S. A., Larsen, S. J., & Rich, C. F. (2016). A primer on potential impacts, management priorities, and future directions for *Elodea* spp. In high latitude systems: Learning from the Alaskan experience. *Hydrobiologia, 777*(1), 1–19. https://doi.org/10.1007/s10750-016-2767-x

Carlson, R. L., Shulman, M. J., & Ellis, J. C. (2006). Factors contributing to spatial heterogeneity in the abundance of the common periwinkle *Littorina littorea* (L.). *Journal of Molluscan Studies, 72*(2), 149–156. https://doi.org/10.1093/mollus/eyi059

Carlton, J. T. (1996). Marine bioinvasions: The alteration of marine ecosystems by nonindigenous species. *Oceanography, 9*(1), 36–43. https://doi.org/10.5670/oceanog.1996.25

Carlton, J. T. (1999a). Molluscan invasions in marine and estuarine communities. *Malacologia, 41*(2), 439–454. https://www.reabic.net/publ/Carlton_1999.pdf

Carlton, J. T. (1999b). The scale and ecological consequences of biological invasions in the World's oceans. In O. T. Sandlund, P. J. Schei, & Å. Viken (Eds.), *Invasive Species and Biodiversity Management* (pp. 195–212). Springer Netherlands. https://doi.org/10.1007/978-94-011-4523-7_13

Carlton, J. T. (2009). Deep invasion ecology and the assembly of communities in historical time. In G. Rilov & J. A. Crooks (Eds.), *Biological Invasions in Marine Ecosystems: Ecological, Management, and Geographic Perspectives* (Vol. 204, pp. 13–56). Springer. https://doi.org/10.1007/978-3-540-79236-9_2

Carlton, J. T. (2011). The Global Dispersal of Marine and Estuarine Crustaceans. In B. S. Galil, P. F. Clark, & J. T. Carlton (Eds.), *In the Wrong Place—Alien Marine Crustaceans: Distribution, Biology and Impacts* (Vol. 6, pp. 3–23). Springer Netherlands. https://doi.org/10.1007/978-94-007-0591-3_1

Carlton, J. T., Chapman, J. W., Geller, J. B., Miller, J. A., Carlton, D. A., McCuller, M. I., Treneman, N. C., Steves, B. P., & Ruiz, G. M. (2017). Tsunami-driven rafting: Transoceanic species dispersal and implications for marine biogeography. *Science, 357*(6358), 1402–1406. https://doi.org/10.1126/science.aao1498

Carlton, J. T., & Eldredge, L. G. (2009). *Marine bioinvasions of Hawai'i: The introduced and cryptogenic marine and estuarine animals and plants of the Hawaiian Archipelago* (Vol. 4). Bernice P. Bishop Museum/Bishop Museum Press.

Carlton, J. T., & Eldredge, L. G. (2015). Update and Revisions of The Marine Bioinvasions of Hawai'i: The Introduced and Cryptogenic Marine and Estuarine Animals and Plants of the Hawaiian Archipelago. *Bishop Museum Bulletin in Zoology, 9*, 25–47. http://hbs.bishopmuseum.org/pubs-online/pdf/bz9-02.pdf

Carlton, J. T., & Fowler, A. E. (2018). Ocean rafting and marine debris: A broader vector menu requires a greater appetite for invasion biology research support. *Aquatic Invasions, 13*(1), 11–15. https://doi.org/10.3391/ai.2018.13.1.02

Carlton, J. T., Keith, I., & Ruiz, G. M. (2019). Assessing marine bioinvasions in the Galápagos Islands: Implications for conservation biology and marine protected areas. *Aquatic Invasions, 14*(1), 1–20. https://doi.org/10.3391/ai.2019.14.1.01

Carnegie, R. B., & Cochennec-Laureau, N. (2004). Microcell parasites of oysters: Recent insights and future trends. *Aquatic Living Resources, 17*(4), 519–528. https://doi.org/10.1051/alr:2004055

Carroll, S. P. (2011). Conciliation biology: The eco-evolutionary management of permanently invaded biotic systems: Conciliation biology. *Evolutionary Applications, 4*(2), 184–199. https://doi.org/10.1111/j.1752-4571.2010.00180.x

Carroll, S. P., Hendry, A. P., Reznick, D. N., & Fox, C. W. (2007). Evolution on ecological time-scales. *Functional Ecology, 21*, 387–393. https://doi.org/10.1111/j.1365-2435.2007.01289.x

Caruso, N., Valenzuela, A. E. J., Burdett, C. L., Luengos Vidal, E. M., Birochio, D., & Casanave, E. B. (2018). Summer habitat use and activity patterns of wild boar *Sus scrofa* in rangelands of central Argentina. *PLoS ONE, 13*(10), e0206513. https://doi.org/10.1371/journal.pone.0206513

Casal, C. M. V. (2006). Global Documentation of Fish Introductions: The Growing Crisis and Recommendations for Action. *Biological Invasions, 8*(1), 3–11. https://doi.org/10.1007/s10530-005-0231-3

Casal, C. M. V., Luna, S., Froese, R., Bailly, N., Atanacio, R., & Agbayani, E. (2007). Alien fish species in the Philippines: Pathways, biological characteristics, establishment and invasiveness. *Journal of Environmental Science and Management, 10*(1), 1–9. https://digitalarchive.worldfishcenter.org/bitstream/handle/20.500.12348/1628/1029.pdf?sequence1=

Cassey, P., Delean, S., Lockwood, J. L., Sadowski, J. S., & Blackburn, T. M. (2018). Dissecting the null model for biological invasions: A meta-analysis of the propagule pressure effect. *PLoS Biology, 16*(4), e2005987. https://doi.org/10.1371/journal.pbio.2005987

Cassey, P., Vall-llosera, M., Dyer, E., & Blackburn, T. M. (2015). 2. The biogeography of avian invasions: History, accident and market trade. In J. Canning-Clode (Ed.), *Biological invasions in changing ecosystems: Vectors, ecological impacts, management and predictions* (pp. 38–54). De Gruyter Open Poland. https://doi.org/10.1515/9783110438666-006

Castellanos-Galindo, G. A., Robertson, D. R., Sharpe, D. M. T., & Torchin, M. E. (2020). A new wave of marine fish invasions through the Panama and Suez canals. *Nature Ecology & Evolution, 4*(11), Article 11. https://doi.org/10.1038/s41559-020-01301-2

Castilla, J. C., & Neill, P. E. (2009). Marine Bioinvasions in the Southeastern Pacific: Status, Ecology, Economic Impacts, Conservation and Management. In G. Rilov & J. A. Crooks (Eds.), *Biological Invasions in Marine Ecosystems: Ecological, Management, and Geographic Perspectives* (Vol. 204, pp. 439–457). Springer. https://doi.org/10.1007/978-3-540-79236-9_26

Castilla, J. C., Uribe, M., Bahamonde, N., Clarke, M., Desqueyroux-Faúndez, R., Kong, I., Moyano, H., Rozbaczylo, N., Santelices, B., Valdovinos, C., & Zavala, P. (2005). Down under the southeastern Pacific: Marine non-indigenous species in Chile. *Biological Invasions, 7*(2), 213–232. https://doi.org/10.1007/s10530-004-0198-5

Catford, J. A., & Jones, L. P. (2019). Grassland invasion in a changing climate. In D. J. Gibson & J. A. Newman (Eds.), *Grasslands and Climate Change* (1st ed., pp. 149–171). Cambridge University Press. https://doi.org/10.1017/9781108163941.011

Catford, J. A., & Kyle, G. (2016). Alien plant invasions in Australia's riparian zones. In S. J. Capon, C. James, & M. Reid (Eds.), *Vegetation of Australian Riverine Landscapes: Biology, Ecology and Management* (pp. 325–342). CSIRO Publishing. http://www.publish.csiro.au/book/6504

Catford, J. A., Naiman, R. J., Chambers, L. E., Roberts, J., Douglas, M., & Davies, P. (2013). Predicting Novel Riparian Ecosystems in a Changing Climate. *Ecosystems, 16*(3), 382–400. https://doi.org/10.1007/s10021-012-9566-7

Catford, J. A., Roberts, J., Capon, S. J., Froend, R. H., Windecker, S. M., & Douglas, M. M. (2017). Wetland vegetation of inland Australia. In D. A. Keith (Ed.), *Australian Vegetation* (pp. 490–515). Cambridge University Press. https://doi.org/10.1017/9781316339701

CBD. (2014). *Pathways of introduction of invasive species, their prioritization and management.* 18. https://www.cbd.int/doc/meetings/sbstta/sbstta-18/official/sbstta-18-09-add1-en.pdf

Celesti-Grapow, L., Alessandrini, A., Arrigoni, P. V., Banfi, E., Bernardo, L., Bovio, M., Brundu, G., Cagiotti, M. R., Camarda, I., Carli, E., Conti, F., Fascetti, S., Galasso, G., Gubellini, L., La Valva, V., Lucchese, F., Marchiori, S., Mazzola, P., Peccenini, S., … Blasi, C. (2009). Inventory of the non-native flora of Italy. *Plant Biosystems – An International Journal Dealing with All Aspects of Plant Biology, 143*(2), 386–430. https://doi.org/10.1080/11263500902722824

Čeplová, N., Lososová, Z., & Kalusová, V. (2017). Urban ornamental trees: A source of current invaders; a case study from a European City. *Urban Ecosystems, 20*(5), 1135–1140. https://doi.org/10.1007/s11252-017-0665-2

Ceschin, S., Abati, S., Ellwood, N. T. W., & Zuccarello, V. (2018). Riding invasion waves: Spatial and temporal patterns of the invasive *Lemna minuta* from its arrival to its spread across Europe. *Aquatic Botany, 150*, 1–8. https://doi.org/10.1016/j.aquabot.2018.06.002

CFS. (2012). *Reported Escapes from Fish Farms 1996-2012.* Center for Food Safety. https://www.centerforfoodsafety.org/files/fish-escapes-chart_14767.pdf

CGAPS. (1997). *The Silent Invasion.* Coordinating Group on Alien Pest Species. http://infografik.com/portfolio/images/tnch/SilentInvasion.pdf

Chacón, E., & Saborío, G. (2012). *Red Interamericana de Información de Especies Invasoras, Costa Rica.* Asociación Para la Conservación y el Estudio de la Biodiversidad.

Chainho, P., Fernandes, A., Amorim, A., Ávila, S. P., Canning-Clode, J., Castro, J. J., Costa, A. C., Costa, J. L., Cruz, T., Gollasch, S., Grazziotin-Soares, C., Melo, R., Micael, J., Parente, M. I., Semedo, J., Silva, T., Sobral, D., Sousa, M., Torres, P., … Costa, M. J. (2015). Non-indigenous species in Portuguese coastal areas, coastal lagoons, estuaries and islands. *Estuarine, Coastal and Shelf Science, 167*, 199–211. https://doi.org/10.1016/j.ecss.2015.06.019

Chambers, R. M., Meyerson, L. A., & Saltonstall, K. (1999). Expansion of *Phragmites australis* into tidal wetlands of North America. *Aquatic Botany, 64*(3), 261–273. https://doi.org/10.1016/S0304-3770(99)00055-8

Chan, F. T., Stanislawczyk, K., Sneekes, A. C., Dvoretsky, A., Gollasch, S., Minchin, D., David, M., Jelmert, A., Albretsen, J., & Bailey, S. A. (2019). Climate change opens new frontiers for marine species in the Arctic: Current trends and future invasion risks. *Global Change Biology, 25*(1), 25–38. https://doi.org/10.1111/gcb.14469

Chapman, D. S., Purse, B. V., Roy, H. E., & Bullock, J. M. (2017). Global trade networks determine the distribution of invasive non-native species. *Global Ecology and Biogeography, 26*(8), 907–917. https://doi.org/10.1111/geb.12599

Chapman, J. W., & Carlton, J. T. (1991). A Test of Criteria for Introduced Species: The Global Invasion by the Isopod *Synidotea Laevidorsalis* (Miers, 1881). *Journal of Crustacean Biology, 11*(3), 386–400. https://doi.org/10.2307/1548465

Chapple, D. G., Knegtmans, J., Kikillus, H., & van Winkel, D. (2016). Biosecurity of exotic reptiles and amphibians in New

Zealand: Building upon Tony Whitaker's legacy. *Journal of the Royal Society of New Zealand, 46*(1), 66–84. https://doi.org/10.1080/03036758.2015.1108344

Cheek, M., Nic Lughadha, E., Kirk, P., Lindon, H., Carretero, J., Looney, B., Douglas, B., Haelewaters, D., Gaya, E., Llewellyn, T., Ainsworth, A. M., Gafforov, Y., Hyde, K., Crous, P., Hughes, M., Walker, B. E., Campostrini Forzza, R., Wong, K. M., & Niskanen, T. (2020). New scientific discoveries: Plants and fungi. *Plants, People, Planet, 2*(5), 371–388. https://doi.org/10.1002/ppp3.10148

Chen, C., Wang, Q.-H., Wu, J.-Y., Huang, D., Zhang, W.-H., Zhao, N., Li, X.-F., & Wang, L.-X. (2017). Historical introduction, geographical distribution, and biological characteristics of alien plants in China. *Biodiversity and Conservation, 26*(2), 353–381. https://doi.org/10.1007/s10531-016-1246-z

Chen, X., Wang, W., Liang, H., Liu, X., & Da, L. (2014). Dynamics of ruderal species diversity under the rapid urbanization over the past half century in Harbin, Northeast China. *Urban Ecosystems, 17*(2), 455–472. https://doi.org/10.1007/s11252-013-0338-8

Chen, Y. (2008). Global potential distribution of an invasive species, the yellow crazy ant *(Anoplolepis gracilipes)* under climate change. *Integrative Zoology, 3*(3), 166–175. https://doi.org/10.1111/j.1749-4877.2008.00095.x

Chen, Y., Sun, C., & Zhan, A. (2017). Biological invasions in aquatic ecosystems in China. *Aquatic Ecosystem Health & Management, 20*(4), 402–412. https://doi.org/10.1080/14634988.2017.1403268

Chinea, J. D., & Helmer, E. H. (2003). Diversity and composition of tropical secondary forests recovering from large-scale clearing: Results from the 1990 inventory in Puerto Rico. *Forest Ecology and Management, 180*(1–3), 227–240. https://doi.org/10.1016/S0378-1127(02)00565-0

Chmura, D. (2020). The spread and role of the invasive alien tree *Quercus rubra* (L.) in novel forest ecosystems in Central Europe. *Forests, 11*(5), 586. https://doi.org/10.3390/f11050586

Chocholoušková, Z., & Pyšek, P. (2003). Changes in composition and structure of urban flora over 120 years: A case study of the city of Plzeň. *Flora – Morphology, Distribution, Functional Ecology of Plants, 198*(5), 366–376. https://doi.org/10.1078/0367-2530-00109

Chomchalow, N. (2011). Giant salvinia—An invasive alien aquatic plant in Thailand. *AU Journal of Technology, 15*(2), 77–82. https://thaiscience.info/Journals/Article/AUJT/10905855.pdf

Chong, K. Y., Corlett, R. T., Nuñez, M. A., Chiu, J. H., Courchamp, F., Dawson, W., Kuebbing, S., Liebhold, A. M., Padmanaba, M., Souza, L., Andersen, K. M., Fei, S., Lee, B. P. Y.-H., Lum, S., Luskin, M. S., Ngo, K. M., & Wardle, D. A. (2021). Are terrestrial biological invasions different in the tropics? *Annual Review of Ecology, Evolution, and Systematics, 52*(1), 291–314. https://doi.org/10.1146/annurev-ecolsys-012021-095454

Chown, S. L., Hodgins, K. A., Griffin, P. C., Oakeshott, J. G., Byrne, M., & Hoffmann, A. A. (2015). Biological invasions, climate change and genomics. *Evolutionary Applications, 8*(1), 23–46. https://doi.org/10.1111/eva.12234

Chown, S. L., Huiskes, A. H. L., Gremmen, N. J. M., Lee, J. E., Terauds, A., Crosbie, K., Frenot, Y., Hughes, K. A., Imura, S., Kiefer, K., Lebouvier, M., Raymond, B., Tsujimoto, M., Ware, C., Van de Vijver, B., & Bergstrom, D. M. (2012). Continent-wide risk assessment for the establishment of nonindigenous species in Antarctica. *Proceedings of the National Academy of Sciences of the United States of America, 109*(13), 4938–4943. https://doi.org/10.1073/pnas.1119787109

Christensen, B., Cashmore, P., Crump, S., & Hobbs, J. (2019). Fire disturbance favours exotic species at Kaituna Wetland, Bay of Plenty. *New Zealand Journal of Ecology, 43*(2), 3369. https://dx.doi.org/10.20417/nzjecol.43.23

Christy, M. T., Clark, C. S., Gee, D. E., Vice, D., Vice, D. S., Warner, M. P., Tyrrell, C. L., Rodda, G. H., & Savidge, J. A. (2007). Recent records of alien anurans on the Pacific Island of Guam. *Pacific Science, 61*(4), 469–483. https://doi.org/10.2984/1534-6188(2007)61[469:rroaao]2.0.co;2

Christy, M. T., Savidge, J. A., & Rodda, G. H. (2007). Multiple pathways for invasion of anurans on a Pacific island. *Diversity and Distributions, 13*(5), 598–607. https://doi.org/10.1111/j.1472-4642.2007.00378.x

Chwedorzewska, K. J., Gielwanowska, I., Olech, M., Molina-Montenegro, M. A., Wodkiewicz, M., & Galera, H. (2015). *Poa annua* L. in the maritime Antarctic: An overview. *POLAR RECORD, 51*(6), 637–643. https://doi.org/10.1017/S0032247414000916

Chwedorzewska, K. J., Korczak-Abshire, M., & Znój, A. (2020). Is Antarctica under threat of alien species invasion? *Global Change Biology, 26*(4), 1942–1943. https://doi.org/10.1111/gcb.15013

Chytrý, M., Maskell, L. C., Pino, J., Pyšek, P., Vilà, M., Font, X., & Smart, S. M. (2008). Habitat invasions by alien plants: A quantitative comparison among Mediterranean, subcontinental and oceanic regions of Europe. *Journal of Applied Ecology, 45*(2), 448–458. https://doi.org/10.1111/j.1365-2664.2007.01398.x

Chytrý, M., Pyšek, P., Wild, J., Pino, J., Maskell, L. C., & Vilà, M. (2009). European map of alien plant invasions based on the quantitative assessment across habitats. *Diversity and Distributions, 15*(1), 98–107. https://doi.org/10.1111/j.1472-4642.2008.00515.x

Chytrý, M., Wild, J., Pyšek, P., Jarošík, V., Dendoncker, N., Reginster, I., Pino, J., Maskell, L. C., Vilà, M., Pergl, J., Kühn, I., Spangenberg, J. H., & Settele, J. (2012). Projecting trends in plant invasions in Europe under different scenarios of future land-use change. *Global Ecology and Biogeography, 21*(1), 75–87. https://doi.org/10.1111/j.1466-8238.2010.00573.x

Cianfanelli, S., Talenti, E., & Bodon, M. (2016). *Mieniplotia scabra* (Müller, 1774), another gastropod invasive species in Europe and the status of freshwater allochthonous molluscs in Greece and Europe. *Mediterranean Marine Science, 17*(1), 253–263. https://doi.org/10.12681/mms.1331

Cilliers, S. S., Williams, N. S. G., & Barnard, F. J. (2008). Patterns of exotic plant invasions in fragmented urban and rural grasslands across continents. *Landscape Ecology, 23*(10), 1243–1256. https://doi.org/10.1007/s10980-008-9295-7

Çinar, M. E., Bilecenoğlu, M., Yokeş, M. B., Öztürk, B., Taşkin, E., Bakir, K., Doğan, A., & Açik, Ş. (2021). Current status (as of end of 2020) of marine alien species in Turkey. *PLoS ONE, 16*(5), e0251086. https://doi.org/10.1371/journal.pone.0251086

Clavero, M., Araujo, R., Calzada, J., Delibes, M., Fernández, N., Gutiérrez-Expósito, C., Revilla, E., & Román, J. (2012). The first invasive bivalve in African fresh waters: Invasion portrait and management options. *Aquatic Conservation: Marine and Freshwater Ecosystems, 22*(2), 277–280. https://doi.org/10.1002/aqc.2231

Clements, E. J., & Foster, M. C. (1994). *Alien plants of the British Isles. A provisional catalogue of vascular plants (excluding grasses)* (p. 589). Botanical Society of the British Isles. https://www.cabi.org/isc/abstract/19952313216

Closset-Kopp, D., Chabrerie, O., Valentin, B., Delachapelle, H., & Decocq, G. (2007). When Oskar meets Alice: Does a lack of trade-off in *r/K*-strategies make *Prunus serotina* a successful invader of European forests? *Forest Ecology and Management, 247*(1–3), 120–130. https://doi.org/10.1016/j.foreco.2007.04.023

Clout, M. N., & Russell, J. C. (2008). The invasion ecology of mammals: A global perspective. *Wildlife Research, 35*(3), 180–184. https://doi.org/10.1071/WR07091

Coetzee, J. A., Hill, M. P., Hussner, A., Nunes, A. L., & Weyl, O. L. F. (2019). Invasive Aquatic Species. In J. M. R. Hughes (Ed.), *Freshwater Ecology and Conservation: Approaches and Techniques* (pp. 338–358). Oxford University Press. https://doi.org/10.1093/oso/9780198766384.003.0016

Coetzee, J. A., Hill, M. P., Ruiz-Téllez, T., Starfinger, U., & Brunel, S. (2017). Monographs on invasive plants in Europe N° 2: *Eichhornia crassipes* (Mart.) Solms. *Botany Letters, 164*(4), 303–326. https://doi.org/10.1080/23818107.2017.1381041

Cogălniceanu, D., Castilla, A. M., Valdeón, A., Gosá, A., Al-Jaidah, N., Alkuwary, A., Saifelnasr, E. O. H., Mas-Peinado, P., Richer, R., & Al-Hemaidi, A. A. M. (2014). A preliminary report on the distribution of lizards in Qatar. *ZooKeys, 373*, 67–91. https://doi.org/10.3897/zookeys.373.5994

Cohen, A. N. (2006). Chapter III Species Introductions and the Panama Canal. In S. Gollasch, B. S. Galil, & A. N. Cohen (Eds.), *Bridging Divides: Maritime Canals as Invasion Corridors* (Vol. 83). Springer. https://doi.org/10.1007/978-1-4020-5047-3_5

Cohen, A. N., & Carlton, J. T. (1998). Accelerating Invasion Rate in a Highly Invaded Estuary. *Science, 279*(5350), 555–558. https://doi.org/10.1126/science.279.5350.555

Colautti, R. I., & Barrett, S. C. H. (2013). Rapid Adaptation to Climate Facilitates Range Expansion of an Invasive Plant. *Science, 342*(6156), 364–366. https://doi.org/10.1126/science.1242121

Colautti, R. I., & Lau, J. A. (2015). Contemporary evolution during invasion: Evidence for differentiation, natural selection, and local adaptation. *Molecular Ecology, 24*(9), 1999–2017. https://doi.org/10.1111/mec.13162

Coleman, J. J. (2016). The *Fusarium solani* species complex: Ubiquitous pathogens of agricultural importance. *Molecular Plant Pathology, 17*(2), 146–158. https://doi.org/10.1111/mpp.12289

Coles, S. L., DeFelice, R. C., Eldredge, L. G., & Carlton, J. T. (1999). Historical and recent introductions of non-indigenous marine species into Pearl Harbor, Oahu, Hawaiian Islands. *Marine Biology, 135*(1), 147–158. https://doi.org/10.1007/s002270050612

Coles, S. L., Reath, P. R., Skelton, P. A., Bonito, V., DeFelice, R. C., & Basch, L. (2003). *Introduced Marine Species in Pago Pago Harbor, Fagatele Bay and the National Park Coast, American Samoa* (26; Bishop Museum Technical Report). Bishop Museum. http://www.botany.hawaii.edu/basch/uhnpscesu/pdfs/sam/Coles2003AS.pdf

Contreras-Balderas, S., Ruiz-Campos, G., Schmitter-Soto, J. J., Díaz-Pardo, E., Contreras-McBeath, T., Medina-Soto, M., Zambrano-González, L., Varela-Romero, A., Mendoza-Alfaro, R., Ramírez-Martínez, C., Leija-Tristán, M. A., Almada-Villela, P., Hendrickson, D. A., & Lyons, J. (2008). Freshwater fishes and water status in México: A country-wide appraisal. *Aquatic Ecosystem Health & Management, 11*(3), 246–256. https://doi.org/10.1080/14634980802319986

Cook, E. J., Ashton, G., Campbell, M., Coutts, A., Gollasch, S., Hewitt, C., Liu, H., Minchin, D., Ruiz, G., & Shucksmith, R. (2008). Non-Native Aquaculture Species Releases: Implications for Aquatic Ecosystems. In M. Holmer, K. Black, C. M. Duarte, N. Marbà, & I. Karakassis (Eds.), *Aquaculture in the Ecosystem*. Springer. https://doi.org/10.1007/978-1-4020-6810-2_5

Correa A., M. D., Galdames, C., & De Stapf, M. S. (2004). *Catálogo de las plantas vasculares de Panamá*. Smithsonian Tropical Research Institute.

Cosham, J., Beazley, K. F., & McCarthy, C. (2016). Environmental factors influencing local distributions of European green crab *(Carcinus maenas)* for modeling and management applications. *Environmental Reviews, 24*(3), 244–252. https://doi.org/10.1139/er-2015-0053

Costanza, R., Fisher, B., Ali, S., Beer, C., Bond, L., Boumans, R., Danigelis, N. L., Dickinson, J., Elliott, C., Farley, J., Gayer, D. E., Glenn, L. M., Hudspeth, T., Mahoney, D., McCahill, L., McIntosh, B., Reed, B., Rizvi, S. A. T., Rizzo, D. M., … Snapp, R. (2006). *Quality of life: An approach integrating opportunities, human needs, and subjective well-being. 61*(2–3), 267–276. https://doi.org/10.1016/j.ecolecon.2006.02.023

Costard, S., Wieland, B., de Glanville, W., Jori, F., Rowlands, R., Vosloo, W., Roger, F., Pfeiffer, D. U., & Dixon, L. K. (2009). African swine fever: How can global spread be prevented? *Philosophical Transactions of the Royal Society B: Biological Sciences, 364*(1530), 2683–2696. https://doi.org/10.1098/rstb.2009.0098

Costello, C. J., & Solow, A. R. (2003). On the pattern of discovery of introduced species. *Proceedings of the National Academy of Sciences, 100*(6), 3321–3323. https://doi.org/10.1073/pnas.0636536100

Costello, K. E., Lynch, S. A., O'Riordan, R. M., McAllen, R., & Culloty, S. C. (2021). The Importance of Marine Bivalves in Invasive Host–Parasite Introductions. *Frontiers in Marine Science, 8*, 147. https://doi.org/10.3389/fmars.2021.609248

Costello, M. J., Basher, Z., Sayre, R., Breyer, S., & Wright, D. J. (2018). Stratifying ocean sampling globally and with depth to account for environmental variability. *Scientific Reports, 8*(1), 11259. https://doi.org/10.1038/s41598-018-29419-1

Costello, M. J., Dekeyzer, S., Galil, B. S., Hutchings, P., Katsanevakis, S., Pagad, S., Robinson, T. B., Turon, X., Vandepitte, L., Vanhoorne, B., Verfaille, K., Willan, R. C., & Rius, M. (2021). Introducing the World Register of Introduced Marine Species (WRiMS). *Management of Biological Invasions, 12*(4), 792–811. https://doi.org/10.3391/mbi.2021.12.4.02

Côté, I. M., Green, S. J., & Hixon, M. A. (2013). Predatory fish invaders: Insights from Indo-Pacific lionfish in the western Atlantic and Caribbean. *Biological Conservation, 164*, 50–61. https://doi.org/10.1016/j.biocon.2013.04.014

Coulson, S. J. (2015). The alien terrestrial invertebrate fauna of the High Arctic archipelago of Svalbard: Potential implications for the native flora and fauna. *Polar Research, 34*(1), 27364. https://doi.org/10.3402/polar.v34.27364

Courchamp, F., Chapuis, J.-L., & Pascal, M. (2003). Mammal invaders on islands:

Impact, control and control impact. *Biological Reviews, 78*(3), 347–383. https://doi.org/10.1017/S1464793102006061

Courchamp, F., Fournier, A., Bellard, C., Bertelsmeier, C., Bonnaud, E., Jeschke, J. M., & Russell, J. C. (2017). Invasion Biology: Specific Problems and Possible Solutions. *Trends in Ecology & Evolution, 32*(1), 13–22. https://doi.org/10.1016/j.tree.2016.11.001

Courtenay, W. R., & Meffe, G. K. (1989). Small fishes in strange places: A review of introduced poeciliids. In G. K. Meffe & F. F. Snelson (Eds.), *Ecology and Evolution of Livebearing Fishes (Poeciliidae)* (pp. 319–331). Prentice-Hall, Inc. https://www.cabdirect.org/cabdirect/abstract/19900502747

Cowie, R. H. (1998). Patterns of introduction of non-indigenous non-marine snails and slugs in the Hawaiian Islands. *Biodiversity and Conservation, 7*(3), 349–368. https://doi.org/10.1023/A:1008881712635

Cowie, R. H. (2005). Alien non-marine molluscs in the islands of the tropical and subtropical Pacific: A review. *American Malacological Bulletin, 20*(1–2), 95–103.

Cowie, R. H., Dillon, R. T., Robinson, D. G., & Smith, J. W. (2009). Alien Non-Marine Snails and Slugs of Priority Quarantine Importance in the United States: A Preliminary Risk Assessment. American Malacological Bulletin, 27(1–2), 113–132. https://doi.org/10.4003/006.027.0210

Cowie, R. H., Hayes, K. A., Tran, C. T., & Meyer, W. M. (2008). The horticultural industry as a vector of alien snails and slugs: Widespread invasions in Hawaii. *International Journal of Pest Management, 54*(4), 267–276. https://doi.org/10.1080/09670870802403986

Cox, G. W. (1999). *Alien Species in North America and Hawaii. Island Press.* https://islandpress.org/books/alien-species-north-america-and-hawaii

Cox, P. A., & Banack, S. A. (1991). *Islands, plants, and Polynesians: An introduction to Polynesian ethnobotany* (p. 228). Dioscorides Press. https://doi.org/10.1016/0378-8741(93)90030-9

Craig, J. F. (1992). Human-induced changes in the composition of fish communities in the African Great Lakes. *Reviews in Fish Biology and Fisheries, 2*(2), 93–124. https://doi.org/10.1007/BF00042880

Cranfield, H. J., Gordon, D. P., Willan, R. C., Marshall, B. A., Battershill, C. N., Francis, M. P., Nelson, W. A., Glasby, C. J., & Read, G. B. (1998). *Adventive marine species in New Zealand* (34; NIWA Technical Report, p. 48). The National Institute of Water and Atmospheric Research. https://docs.niwa.co.nz/library/public/NIWAtr34.pdf

Crawford, S. S., & Muir, A. M. (2008). Global introductions of salmon and trout in the genus *Oncorhynchus*: 1870–2007. *Reviews in Fish Biology and Fisheries, 18*(3), 313–344. https://doi.org/10.1007/s11160-007-9079-1

Croce, M. E., & Parodi, E. R. (2014). The Japanese alga *Polysiphonia morrowii* (Rhodomelaceae, Rhodophyta) on the South Atlantic Ocean: First report of an invasive macroalga inhabiting oyster reefs. *Helgoland Marine Research, 68*(2), 241–252. https://doi.org/10.1007/s10152-014-0384-5

Cronin, J. T., Bhattarai, G. P., Allen, W. J., & Meyerson, L. A. (2015). Biogeography of a plant invasion: Plant—herbivore interactions. *Ecology, 96*(4), 1115–1127. https://www.jstor.org/stable/43494769

Crooks, J. A. (2005). Lag times and exotic species: The ecology and management of biological invasions in slow-motion1. *Écoscience, 12*(3), 316–329. https://doi.org/10.2980/i1195-6860-12-3-316.1

Crooks, J. A., & Khim, H. S. (1999). Architectural vs. Biological effects of a habitat-altering, exotic mussel, *Musculista senhousia. Journal of Experimental Marine Biology and Ecology, 240*(1), 53–75. https://doi.org/10.1016/S0022-0981(99)00041-6

Crous, P. W., Rong, I. H., Wood, A., Lee, S., Glen, H., Botha, W., Slippers, B., de Beer, W. Z., Wingfield, M. J., & Hawksworth, D. L. (2006). How many species of fungi are there at the tip of Africa? *Studies in Mycology, 55*, 13–33. https://doi.org/10.3114/sim.55.1.13

Crowley, S. L. (2014). Camels Out of Place and Time: The Dromedary *(Camelus dromedarius)* in Australia. *Anthrozoös, 27*(2), 191–203. https://doi.org/10.2752/175303714X13903827487449

Crystal-Ornelas, R., & Lockwood, J. L. (2020). The 'known unknowns' of invasive species impact measurement. *Biological Invasions, 22*(4), 1513–1525. https://doi.org/10.1007/s10530-020-02200-0

Csuzdi, C., Chang, C.-H., Pavlícek, T., Szederjesi, T., Esopi, D., & Szlávecz, K. (2017). Molecular phylogeny and systematics of native North American lumbricid earthworms (Clitellata: Megadrili). *PLoS ONE, 12*(8), e0181504. https://doi.org/10.1371/journal.pone.0181504

Cucherousset, J., Blanchet, S., & Olden, J. D. (2012). Non-native species promote trophic dispersion of food webs. *Frontiers in Ecology and the Environment, 10*(8), 406–408. https://doi.org/10.1890/12.WB.018

Cusack, D. F., & McCleery, T. L. (2014). Patterns in understory woody diversity and soil nitrogen across native- and non-native-urban tropical forests. *Forest Ecology and Management, 318*, 34–43. https://doi.org/10.1016/j.foreco.2013.12.036

Cuthbert, R. N., Kotronaki, S. G., Dick, J. T. A., & Briski, E. (2020). Salinity tolerance and geographical origin predict global alien amphipod invasions. *Biology Letters, 16*(9), 20200354. https://doi.org/10.1098/rsbl.2020.0354

da Rosa, C. A., Zenni, R., Ziller, S. R., de Almeida Curi, N., & Passamani, M. (2018). Assessing the risk of invasion of species in the pet trade in Brazil. *Perspectives in Ecology and Conservation, 16*(1), 38–42. https://doi.org/10.1016/j.pecon.2017.09.005

Daehler, C. C., & Strong, D. R. (1997). Hybridization between introduced smooth cordgrass (Spartina alternifolia; Poaceae) and native California cordgrass *(S. foliosa)* in San Francisco Bay, California, USA. *American Journal of Botany, 84*(5), 607–611. https://doi.org/10.2307/2445896

Dainese, M., Aikio, S., Hulme, P. E., Bertolli, A., Prosser, F., & Marini, L. (2017). Human disturbance and upward expansion of plants in a warming climate. *Nature Climate Change, 7*(8), 577–580. https://doi.org/10.1038/nclimate3337

Dale, A. G., & Frank, S. D. (2017). Warming and drought combine to increase pest insect fitness on urban trees. *PLoS ONE, 12*(3), e0173844. https://doi.org/10.1371/journal.pone.0173844

D'Amen, M., & Azzurro, E. (2020). Lessepsian fish invasion in Mediterranean marine protected areas: A risk assessment under climate change scenarios. *ICES Journal of Marine Science, 77*(1), 388–397. https://doi.org/10.1093/icesjms/fsz207

D'Antonio, C. M., & Vitousek, P. M. (1992). Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change. *Annual Review of Ecology and Systematics, 23*, 63–87. JSTOR. https://doi.org/annurev.es.23.110192.000431

Dar, J. A., Subashree, K., Sundarapandian, S., Saikia, P., Kumar, A., Khare, P. K., Dayanandan, S., & Khan, M. L. (2019). Invasive Species and Their Impact on Tropical Forests of Central India: A Review. In S. C. Garkoti, S. J. Van Bloem, P. Z. Fulé, & R. L. Semwal (Eds.), *Tropical Ecosystems: Structure, Functions and Challenges in the Face of Global Change* (pp. 69–109). Springer. https://doi.org/10.1007/978-981-13-8249-9_5

Darrigran, G., Agudo-Padrón, I., Baez, P., Belz, C., Cardoso, F., Carranza, A., Collado, G., Correoso, M., Cuezzo, M. G., Fabres, A., Gutiérrez Gregoric, D. E., Letelier, S., Ludwig, S., Mansur, M. C., Pastorino, G., Penchaszadeh, P., Peralta, C., Rebolledo, A., Rumi, A., … Damborenea, C. (2020). Non-native mollusks throughout South America: Emergent patterns in an understudied continent. *Biological Invasions, 22*(3), 853–871. https://doi.org/10.1007/s10530-019-02178-4

Darwall, W. R. T., Smith, K. G., Allen, D. J., Holland, R. A., Harrison, I. J., & Brooks, E. G. E. (2011). *The diversity of life in African fresh-waters: Underwater, under threat. An analysis of the status and distribution of freshwater species throughout mainland Africa.* IUCN.

Das, I. (2015). *A Field Guide to the Reptiles of South-East Asia*. Bloomsbury Publishing.

Daskalov, G. M., Grishin, A. N., Rodionov, S., & Mihneva, V. (2007). Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. *Proceedings of the National Academy of Sciences of the United States of America, 104*(25), 10518–10523. https://doi.org/10.1073/pnas.0701100104

Daskalov, G. M., & Mamedov, E. V. (2007). Integrated fisheries assessment and possible causes for the collapse of anchovy kilka in the Caspian Sea. *ICES Journal of Marine Science, 64*(3), 503–511. https://doi.org/10.1093/icesjms/fsl047

Dassou, A. G., Idohou, R., Azandémè-Hounmalon, G. Y., Sabi-Sabi, A., Houndété, J., Silvie, P., & Dansi, A. (2021). Fall armyworm, *Spodoptera frugiperda* (J.E. Smith) in maize cropping systems in Benin: Abundance, damage, predatory ants and potential control. *International Journal of Tropical Insect Science, 41*(4), 2627–2636. https://doi.org/10.1007/s42690-021-00443-5

Davenport, T. R. B., Fakih, S. A., Kimiti, S. P., Kleine, L. U., Foley, L. S., & De Luca, D. W. (2019). Zanzibar's endemic red colobus *Piliocolobus kirkii*: First systematic and total assessment of population, demography and distribution. *Oryx, 53*(1), 36–44. https://doi.org/10.1017/S003060531700148X

Davidson, A. M., Jennions, M., & Nicotra, A. B. (2011). Do invasive species show higher phenotypic plasticity than native species and, if so, is it adaptive? A meta-analysis: Invasive species have higher phenotypic plasticity. *Ecology Letters, 14*(4), 419–431. https://doi.org/10.1111/j.1461-0248.2011.01596.x

Davidson, C. B., Gottschalk, K. W., & Johnson, E. (1999). Tree Mortality Following Defoliation by the European Gypsy Moth (*Lymantria dispar* L.). In the. United States: A Review. *Forest Science, 45*(1), 74–84. https://doi.org/10.1093/forestscience/45.1.74

Davies, D., Dilley, B. J., Bond, A. L., Cuthbert, R. J., & Ryan, P. G. (2015). Trends and tactics of mouse predation on Tristan Albatross *Diomedea dabbenena* chicks at Gough Island, South Atlantic Ocean. *Avian Conservation and Ecology, 10*(1), art5. https://doi.org/10.5751/ACE-00738-100105

Davis, A. J. S., & Darling, J. A. (2017). Recreational freshwater fishing drives non-native aquatic species richness patterns at a continental scale. *Diversity and Distributions, 23*(6), 692–702. https://doi.org/10.1111/ddi.12557

Davis-Berg, E. C. (2012). The predatory snail *Euglandina rosea* successfully follows mucous trails of both native and non-native prey snails. *Invertebrate Biology, 131*(1), Article 1. https://doi.org/10.1111/j.1744-7410.2011.00251.x

Dawson, W., Moser, D., van Kleunen, M., Kreft, H., Pergl, J., Pyšek, P., Weigelt, P., Winter, M., Lenzner, B., Blackburn, T. M., Dyer, E. E., Cassey, P., Scrivens, S. L., Economo, E. P., Guénard, B., Capinha, C., Seebens, H., García-Díaz, P., Nentwig, W., … Essl, F. (2017). Global hotspots and correlates of alien species richness across taxonomic groups. *Nature Ecology & Evolution, 1*(7), 0186. https://doi.org/10.1038/s41559-017-0186

Day, R., Abrahams, P., Bateman, M., Beale, T., Clottey, V., Cock, M., Colmenarez, Y., Corniani, N., Early, R., Godwin, J., Gomez, J., Moreno, P. G., Murphy, S. T., Oppong-Mensah, B., Phiri, N., Pratt, C., Silvestri, S., & Witt, A. (2017). Fall Armyworm: Impacts and Implications for Africa. *Outlooks on Pest Management, 28*(5), 196–201. https://doi.org/10.1564/v28_oct_02

De Abreu, R. C. R., & Durigan, G. (2011). Changes in the plant community of a Brazilian grassland savannah after 22 years of invasion by *Pinus elliottii* Engelm. *Plant Ecology & Diversity, 4*(2–3), 269–278. https://doi.org/10.1080/17550874.2011.594101

De Clerck, O., Guiry, M. D., Leliaert, F., Samyn, Y., & Verbruggen, H. (2013). Algal taxonomy: A road to nowhere? *Journal of Phycology, 49*(2), Article 2. https://doi.org/10.1111/jpy.12020

De Francesco, C. G., & Lagiglia, H. (2007). A predatory land snail invades central-western Argentina. *Biological Invasions, 9*(7), 795–798. https://doi.org/10.1007/s10530-006-9076-7

De Groote, H., Kimenju, S. C., Munyua, B., Palmas, S., Kassie, M., & Bruce, A. (2020). Spread and impact of fall armyworm (*Spodoptera frugiperda* J.E. Smith) in maize production areas of Kenya. *Agriculture, Ecosystems and Environment, 292*. https://doi.org/10.1016/j.agee.2019.106804

De Lourdes Torres, M., & Mena, C. F. (Eds.). (2018). *Understanding Invasive Species in the Galapagos Islands: From the Molecular to the Landscape* (1st ed. 2018). Springer International Publishing : Imprint: Springer. https://doi.org/10.1007/978-3-319-67177-2

De Menezes, J., & Pagad, S. (2020). *Global Register of Introduced and Invasive Species- Sao Tome and Principe* [dataset]. Invasive Species Specialist Group ISSG. https://doi.org/10.15468/GE0TTB

De Pooter, D., Appeltans, W., Bailly, N., Bristol, S., Deneudt, K., Eliezer, M., Fujioka, E., Giorgetti, A., Goldstein, P., Lewis, M., Lipizer, M., Mackay, K., Marin, M., Moncoiffé, G., Nikolopoulou, S., Provoost, P., Rauch, S., Roubicek, A., Torres, C., … Hernandez, F. (2017). Toward a new data standard for combined marine biological and environmental datasets—Expanding OBIS beyond species occurrences. *Biodiversity Data Journal, 5*, e10989. https://doi.org/10.3897/BDJ.5.e10989

De Silva, S. S. (2012). Aquaculture: A newly emergent food production sector—and perspectives of its impacts on biodiversity and conservation. *Biodiversity and Conservation, 21*(12), 3187–3220. https://doi.org/10.1007/s10531-012-0360-9

De Silva, S. S., Nguyen, T. T. T., Abery, N. W., & Amarasinghe, U. S. (2006). An evaluation of the role and impacts of alien finfish in Asian inland aquaculture.

Aquaculture Research, 37(1), 1–17. https://doi.org/10.1111/j.1365-2109.2005.01369.x

De Silva, S. S., Nguyen, T. T. T., Turchini, G. M., Amarasinghe, U. S., & Abery, N. W. (2009). Alien Species in Aquaculture and Biodiversity: A Paradox in Food Production. *AMBIO: A Journal of the Human Environment, 38*(1), 24–28. https://doi.org/10.1579/0044-7447-38.1.24

Dean, R., Van Kan, J. A. L., Pretorius, Z. A., Hammond-Kosack, K. E., Di Pietro, A., Spanu, P. D., Rudd, J. J., Dickman, M., Kahmann, R., Ellis, J., & Foster, G. D. (2012). The Top 10 fungal pathogens in molecular plant pathology: Top 10 fungal pathogens. *Molecular Plant Pathology, 13*(4), 414–430. https://doi.org/10.1111/j.1364-3703.2011.00783.x

Dehnen-Schmutz, K., Touza, J., Perrings, C., & Williamson, M. (2007). The horticultural trade and ornamental plant invasions in Britain. *Conservation Biology, 21*(1), 224–231. https://doi.org/10.1111/j.1523-1739.2006.00538.x

DeKeyser, E. S., Dennhardt, L. A., & Hendrickson, J. (2015). Kentucky bluegrass *(Poa pratensis)* Invasion in the Northern Great Plains: A Story of Rapid Dominance in an Endangered Ecosystem. *Invasive Plant Science and Management, 8*(3), 255–261. Cambridge Core. https://doi.org/10.1614/IPSM-D-14-00069.1

DeKeyser, S., Meehan, M., Sedivec, K., & Lura, C. (2010). Potential Management Alternatives for Invaded Rangelands in the Northern Great Plains. *Rangelands, 32*(5), 26–31. https://doi.org/10.2111/RANGELANDS-D-10-00055.1

Dembitzer, J. (2017). The Case for Hippos in Colombia. *Israel Journal of Ecology and Evolution, 63*(3–4), 5–8. https://doi.org/10.1163/22244662-06303002

Denslow, J. S., Space, J. C., & Thomas, P. A. (2009). Invasive Exotic Plants in the Tropical Pacific Islands: Patterns of Diversity. *Biotropica, 41*(2), 162–170. https://doi.org/10.1111/j.1744-7429.2008.00469.x

Desprez-Loustau, M.-L. (2009). Alien Fungi of Europe. In *Handbook of Alien Species in Europe* (Vol. 3, pp. 15–28). Springer Netherlands. https://doi.org/10.1007/978-1-4020-8280-1_2

Desprez-Loustau, M.-L., Courtecuisse, R., Robin, C., Husson, C., Moreau, P.-A., Blancard, D., Selosse, M.-A., Lung-Escarmant, B., Piou, D., & Sache, I. (2010). Species diversity and drivers of spread of alien fungi *(sensu lato)* in Europe with a particular focus on France. *Biological Invasions, 12*(1), 157–172. https://doi.org/10.1007/s10530-009-9439-y

Desprez-Loustau, M.-L., Robin, C., Buée, M., Courtecuisse, R., Garbaye, J., Suffert, F., Sache, I., & Rizzo, D. (2007). The fungal dimension of biological invasions. *Trends in Ecology & Evolution, 22*(9), 472–480. https://doi.org/10.1016/j.tree.2007.04.005

Di Castri, F. (1989). History of biological invasions with special emphasis on the Old World. In *Biological Invasion: A Global Perspective* (pp. 1–30). Wiley.

Dibble, K. L., & Meyerson, L. A. (2014). The Effects of Plant Invasion and Ecosystem Restoration on Energy Flow through Salt Marsh Food Webs. *Estuaries and Coasts, 37*(2), 339–353. https://doi.org/10.1007/s12237-013-9673-5

DIISE. (2020). *The Database of Island Invasive Species Eradications, developed by Island Conservation.* Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand. http://diise.islandconservation.org/

Dittel, A. I., & Epifanio, C. E. (2009). Invasion biology of the Chinese mitten crab *Eriochier sinensis*: A brief review. *Journal of Experimental Marine Biology and Ecology, 374*(2), 79–92. https://doi.org/10.1016/j.jembe.2009.04.012

Dixon, A. P., Faber-Langendoen, D., Josse, C., Morrison, J., & Loucks, C. J. (2014). Distribution mapping of world grassland types. *Journal of Biogeography, 41*(11), 2003–2019. https://doi.org/10.1111/jbi.12381

Dlugosch, K. M., & Parker, I. M. (2008). Invading populations of an ornamental shrub show rapid life history evolution despite genetic bottlenecks. *Ecology Letters, 11*, 701–709. https://doi.org/10.1111/j.1461-0248.2008.01181.x

Dodds, K. J., & Orwig, D. A. (2011). An invasive urban forest pest invades natural environments—Asian longhorned beetle in northeastern US hardwood forests. *Canadian Journal of Forest Research, 41*(9), 1729–1742. https://doi.org/10.1139/x11-097

Doi, W., Watanabe, S., & Carlton, J. T. (2011). Alien Marine Crustaceans of Japan: A Preliminary Assessment. In B. S. Galil, P. F. Clark, & J. T. Carlton (Eds.), *In the Wrong Place—Alien Marine Crustaceans: Distribution, Biology and Impacts* (Vol. 6, pp. 419–449). Springer Netherlands. https://doi.org/10.1007/978-94-007-0591-3_15

Dölle, K., & Kurzmann, D. E. (2020). The Freshwater Mollusk *Dreissena polymorpha* (Zebra Mussel)—A Review: Living, Prospects and Jeopardies. *Asian Journal of Environment & Ecology, 13*(2), 1–17. https://doi.org/10.9734/ajee/2020/v13i230176

Dong, X., Ju, T., Grenouillet, G., Laffaille, P., Lek, S., & Liu, J. (2020). Spatial pattern and determinants of global invasion risk of an invasive species, sharpbelly *Hemiculter leucisculus* (Basilesky, 1855). *Science of The Total Environment, 711.* https://doi.org/10.1016/j.scitotenv.2019.134661

Dorcas, M. E., Willson, J. D., Reed, R. N., Snow, R. W., Rochford, M. R., Miller, M. A., Meshaka, W. E., Andreadis, P. T., Mazzotti, F. J., Romagosa, C. M., & Hart, K. M. (2012). Severe mammal declines coincide with proliferation of invasive Burmese pythons in Everglades National Park. *Proceedings of the National Academy of Sciences, 109*(7), 2418–2422. https://doi.org/10.1073/pnas.1115226109

Doria, C. R. da C., Agudelo, E., Akama, A., Barros, B., Bonfim, M., Carneiro, L., Briglia-Ferreira, S. R., Nobre Carvalho, L., Bonilla-Castillo, C. A., Charvet, P., dos Santos Catâneo, D. T. B., da Silva, H. P., Garcia-Dávila, C. R., dos Anjos, H. D. B., Duponchelle, F., Encalada, A., Fernandes, I., Florentino, A. C., Guarido, P. C. P., … Vitule, J. R. S. (2021). The Silent Threat of Non-native Fish in the Amazon: ANNF Database and Review. *Frontiers in Ecology and Evolution, 9*, 646702. https://doi.org/10.3389/fevo.2021.646702

Doria, C. R. da C., Catâneo, D. T. B. dos S., Torrente-Vilara, G., & Vitule, J. R. S. (2020). Is there a future for artisanal fishing in the Amazon? The case of *Arapaima gigas. Management of Biological Invasions, 11*(1), 1–8. https://doi.org/10.3391/mbi.2020.11.1.01

Dorjee, Johnson, S. B., Buckmaster, A. J., & Downey, P. O. (2020). Weeds in the land of Gross National Happiness: Knowing what to manage by creating a baseline alien plant inventory for Bhutan. *Biological Invasions, 22*, 2899–2914. https://doi.org/10.1007/s10530-020-02306-5

dos Santos, D. A., de Paiva Affonso, I., Message, H. J., Okada, E. K., Gomes, L. C., Bornatowski, H., & Vitule, J. R. S. (2019). Societal perception, impacts and judgment values about invasive freshwater stingrays. *Biological Invasions, 21*(12), 3593–3606. https://doi.org/10.1007/s10530-019-02071-0 dos Santos, L. A., Mendes, M. F., Krüger, A. P., Blauth, M. L., Gottschalk, M. S., & Garcia, F. R. M. (2017). Global potential distribution of *Drosophila suzukii* (Diptera, Drosophilidae). *PLoS ONE, 12*(3), e0174318. https://doi.org/10.1371/journal.pone.0174318

Dowell, R. V., Gill, R. J., Jeske, D. R., & Hoddle, M. S. (2016). Exotic Terrestrial Macro-Invertebrate Invaders in California from 1700 to 2015: An Analysis of Records. *Proceedings of the California Academy of Sciences, 63*(3), 63–157. https://biostor.org/reference/249103

Downs, C. T., & Coates, G. D. (2005). Survey of the status and management of sympatric bushbuck and nyala in KwaZulu-Natal, South Africa: Research article. *South African Journal of Wildlife Research – 24-Month Delayed Open Access, 35*(2), 179–190. https://doi.org/10.10520/EJC117215

Drake, J. A., Mooney, H. A., di Castri, F., Groves, R. H., Kruger, F. J., Rejmánek, M., & Williamson, M. (Eds.). (1989). *Biological Invasions: A Global Perspective*. Wiley.

Dreborg, K. H. (1996). Essence of backcasting. *Futures, 28*(9), 813–828. https://doi.org/10.1016/S0016-3287(96)00044-4

Dresseno, A. L. P., Guido, A., Balogianni, V., & Overbeck, G. E. (2018). Negative effects of an invasive grass, but not of native grasses, on plant species richness along a cover gradient. *Austral Ecology, 43*(8), 949–954. https://doi.org/10.1111/aec.12644

Dubey, S. C., Gupta, K., Akhtar, J., Chalam, V. C., Singh, M. C., Khan, Z., Singh, S. P., Kumar, P., Gawade, B. H., Kiran, R., Boopathi, T., & Kumari, P. (2021). Plant quarantine for biosecurity during transboundary movement of plant genetic resources. *Indian Phytopathology, 74*(2), 495–508. https://doi.org/10.1007/s42360-021-00375-7

Dudley, J. P., Hoberg, E. P., Jenkins, E. J., & Parkinson, A. J. (2015). Climate Change in the North American Arctic: A One Health Perspective. *EcoHealth, 12*, 713–725. https://doi.org/10.1007/s10393-015-1036-1

Duffy, G. A., Coetzee, B. W. T., Latombe, G., Akerman, A. H., McGeoch, M. A., & Chown, S. L. (2017). Barriers to globally invasive species are weakening across the Antarctic. *Diversity and Distributions, 23*(9), 982–996. https://doi.org/10.1111/ddi.12593

Dullinger, I., Wessely, J., Bossdorf, O., Dawson, W., Essl, F., Gattringer, A., Klonner, G., Kreft, H., Kuttner, M., Moser, D., Pergl, J., Pyšek, P., Thuiller, W., van Kleunen, M., Weigelt, P., Winter, M., & Dullinger, S. (2017). Climate change will increase the naturalization risk from garden plants in Europe. *Global Ecology and Biogeography, 26*(1), 43–53. https://doi.org/10.1111/geb.12512

Dunn, P. H. (1985). Origins of leafy spurge in North America. In A. K. Watson (Ed.), *Leafy Spurge* (pp. 7–13). Weed Science Society of America.

Durán, A., Gryzenhout, M., Slippers, B., Ahumada, R., Rotella, A., Flores, F., Wingfield, B. D., & Wingfield, M. J. (2008). *Phytophthora pinifolia* sp. Nov. Associated with a serious needle disease of *Pinus radiata* in Chile. *Plant Pathology, 57*(4), 715–727. https://doi.org/10.1111/j.1365-3059.2008.01893.x

Dutton, J. (1994). Introduced mammals in São Tomé and Príncipe: Possible threats to biodiversity. *Biodiversity & Conservation, 3*(9), 927–938. https://doi.org/10.1007/BF00129667

Dvoretsky, A. G., & Dvoretsky, V. G. (2018). Red king crab *(Paralithodes camtschaticus)* fisheries in Russian waters: Historical review and present status. *Reviews in Fish Biology and Fisheries, 28*(2), 331–353. https://doi.org/10.1007/s11160-017-9510-1

Dyer, E. E., Cassey, P., Redding, D. W., Collen, B., Franks, V., Gaston, K. J., Jones, K. E., Kark, S., Orme, C. D. L., & Blackburn, T. M. (2017). The global distribution and drivers of alien bird species richness. *PLoS Biology, 15*(1), e2000942. https://doi.org/10.1371/journal.pbio.2000942

Dyer, E. E., Redding, D. W., & Blackburn, T. M. (2017). The global avian invasions atlas, a database of alien bird distributions worldwide. *Scientific Data, 4*(1), 170041. https://doi.org/10.1038/sdata.2017.41

Dyer, M. J. B., Keppel, G., Tuiwawa, M., Vido, S., & Boehmer, H. J. (2018). Invasive alien palm *Pinanga coronata* threatens native tree ferns in an oceanic island rainforest. *Australian Journal of Botany, 66*(8), 647–656. https://doi.org/10.1071/BT18088

Early, R., Bradley, B. A., Dukes, J. S., Lawler, J. J., Olden, J. D., Blumenthal, D. M., Gonzalez, P., Grosholz, E. D., Ibañez, I., Miller, L. P., Sorte, C. J. B., & Tatem, A. J. (2016). Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications, 7*(1), 12485. https://doi.org/10.1038/ncomms12485

Early, R., González-Moreno, P., Murphy, S. T., & Day, R. (2018). Forecasting the global extent of invasion of the cereal pest *Spodoptera frugiperda*, the fall armyworm. *NeoBiota, 40*, 25–50. https://doi.org/10.3897/neobiota.40.28165

Ebenhard, T. (1988). Introduced birds and mammals and their ecological effects. *Swedish Wildlife Research (Sweden), 13*(4), 1–107. https://scholar.google.com/scholar_lookup?title=Introduced+birds+and+mammals+and+their+ecological+effects&author=Ebenhard%2C+T.+%28Uppsala+Univ.+%28Sweden%29.+Zoologiska+Inst.%29&publication_year=1988

Egoh, B. N., Ntshotsho, P., Maoela, M. A., Blanchard, R., Ayompe, L. M., & Rahlao, S. (2020). Setting the scene for achievable post-2020 convention on biological diversity targets: A review of the impacts of invasive alien species on ecosystem services in Africa. *Journal of Environmental Management, 261*, 110171. https://doi.org/10.1016/j.jenvman.2020.110171

Ehbrecht, M., Seidel, D., Annighöfer, P., Kreft, H., Köhler, M., Zemp, D. C., Puettmann, K., Nilus, R., Babweteera, F., Willim, K., Stiers, M., Soto, D., Boehmer, H. J., Fisichelli, N., Burnett, M., Juday, G., Stephens, S. L., & Ammer, C. (2021). Global patterns and climatic controls of forest structural complexity. *Nature Communications, 12*(1), 519. https://doi.org/10.1038/s41467-020-20767-z

Eijsackers, H. (2011). Earthworms as colonizers of natural and cultivated soil environments. *Applied Soil Ecology, 50*, 1–13. https://doi.org/10.1016/j.apsoil.2011.07.008

Elderd, B. D. (2003). The Impact of Changing Flow Regimes on Riparian Vegetation and the Riparian Species *Mimulus Guttatus. Ecological Applications, 13*(6), 1610–1625. https://doi.org/10.1890/02-5371

Eldredge, L. G. (1994). *Perspectives in Aquatic Exotic Species Management in the Pacific Islands: Introductions of commercially significant aquatic organisms to the Pacific Islands* (Vol. 1). South Pacific Commission. https://purl.org/spc/digilib/doc/qedxg

Ellender, B. R., & Weyl, O. L. F. (2014). A review of current knowledge, risk and ecological impacts associated with non-native freshwater fish introductions in South

Africa. *Aquatic Invasions, 9*(2), 117–132. https://doi.org/10.3391/ai.2014.9.2.01

Ellstrand, N. C., & Schierenbeck, K. A. (2000). Hybridization as a stimulus for the evolution of invasiveness in plants? *Proceedings of the National Academy of Sciences, 97*(13), 7043–7050. https://doi.org/10.1073/pnas.97.13.7043

Eminniyaz, A., Qiu, J., Baskin, C. C., Baskin, J. M., & Tan, D. (2017). Biological Invasions in Desert Green-Islands and Grasslands. In F. Wan, M. Jiang, & A. Zhan (Eds.), *Biological Invasions and Its Management in China: Volume 1* (Vol. 11, pp. 97–123). Springer Netherlands. https://doi.org/10.1007/978-94-024-0948-2_5

Engeman, R. M., Shiels, A. B., & Clark, C. S. (2018). Objectives and integrated approaches for the control of brown tree snakes: An updated overview. *Journal of Environmental Management, 219,* 115–124. https://doi.org/10.1016/j.jenvman.2018.04.092

Enríquez, N., Pertierra, L. R., Tejedo, P., Benayas, J., Greenslade, P., & Luciáñez, M. J. (2019). The importance of long-term surveys on species introductions in Maritime Antarctica: First detection of *Ceratophysella succinea* (Collembola: Hypogastruridae). *Polar Biology, 42*, 1047–1051. https://doi.org/10.1007/s00300-019-02490-8

Epifanio, C. E. (2013). Invasion biology of the Asian shore crab *Hemigrapsus sanguineus*: A review. *Journal of Experimental Marine Biology and Ecology, 441*, 33–49. https://doi.org/10.1016/j.jembe.2013.01.010

EPPO. (2021). *European and Mediterranean Plant Protection Organization (Organisation Européenne et Méditerranéenne pour la Protection des Plantes)*. https://www.eppo.int/

Érsek, T., & Nagy, Z. Á. (2008). Species hybrids in the genus *Phytophthora* with emphasis on the alder pathogen *Phytophthora alni*: A review. In A. Lebeda, P. T. N. Spencer-Phillips, & B. M. Cooke (Eds.), *The Downy Mildews—Genetics, Molecular Biology and Control* (pp. 31–39). Springer Netherlands. https://doi.org/10.1007/978-1-4020-8973-2_4

Esler, A. E. (1987). The naturalisation of plants in urban Auckland, New Zealand 1. The introduction and spread of alien plants. *New Zealand Journal of Botany, 25*(4), 511–522. https://doi.org/10.1080/0028825X.1987.10410083

Essl, F., Bacher, S., Genovesi, P., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kowarik, I., Kühn, I., Pyšek, P., Rabitsch, W., Schindler, S., van Kleunen, M., Vilà, M., Wilson, J. R. U., & Richardson, D. M. (2018). Which taxa are alien? Criteria, applications, and uncertainties. *BioScience, 68*(7), 496–509. https://doi.org/10.1093/biosci/biy057

Essl, F., Dawson, W., Kreft, H., Pergl, J., Pyšek, P., Van Kleunen, M., Weigelt, P., Mang, T., Dullinger, S., Lenzner, B., Moser, D., Maurel, N., Seebens, H., Stein, A., Weber, E., Chatelain, C., Inderjit, Genovesi, P., Kartesz, J., … Winter, M. (2019). Drivers of the relative richness of naturalized and invasive plant species on Earth. *AoB PLANTS, 11*(5), plz051. https://doi.org/10.1093/aobpla/plz051

Essl, F., Lenzner, B., Bacher, S., Bailey, S., Capinha, C., Daehler, C., Dullinger, S., Genovesi, P., Hui, C., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kühn, I., Leung, B., Liebhold, A., Liu, C., MacIsaac, H. J., Meyerson, L. A., Nuñez, M. A., … Roura-Pascual, N. (2020). Drivers of future alien species impacts: An expert-based assessment. *Global Change Biology, 26*(9), 4880–4893. https://doi.org/10.1111/gcb.15199

Essl, F., Milasowszky, N., & Dirnböck, T. (2011). Plant invasions in temperate forests: Resistance or ephemeral phenomenon? *Basic and Applied Ecology, 12*(1), 1–9. https://doi.org/10.1016/j.baae.2010.10.003

Essl, F., & Rabitsch, W. (2002). *Neobiota in Österreich* (p. 432). Federal Environment Agency – Austria. https://www.umweltbundesamt.at/fileadmin/site/publikationen/dp089.pdf

Essl, F., Steinbauer, K., Dullinger, S., Mang, T., & Moser, D. (2013). Telling a different story: A global assessment of bryophyte invasions. *Biological Invasions, 15*(9), Article 9. https://doi.org/10.1007/s10530-013-0422-2

Estoup, A., & Guillemaud, T. (2010). Reconstructing routes of invasion using genetic data: Why, how and so what? *Molecular Ecology, 19*(19), 4113–4130. https://doi.org/10.1111/j.1365-294X.2010.04773.x

European Union. (2014). *Regulation (EU) No 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species.* Official Journal of the European Union. http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32014R1143&from=EN

Evans, T. (2021). Quantifying the global threat to native birds from predation by non-native birds on small islands. *Conservation Biology, 35*(4), 1268–1277. https://doi.org/10.1111/cobi.13697

Evans, T., Kumschick, S., & Blackburn, T. M. (2016). Application of the Environmental Impact Classification for Alien Taxa (EICAT) to a global assessment of alien bird impacts. *Diversity and Distributions, 22*(9), 919–931. https://doi.org/10.1111/ddi.12464

Eviner, V. T., Garbach, K., Baty, J. H., & Hoskinson, S. A. (2012). Measuring the effects of invasive plants on ecosystem services: Challenges and prospects. *Invasive Plant Science and Management, 5*(1), 125–136. https://doi.org/10.1614/IPSM-D-11-00095.1

Faccia, I., Alyakrinsky, A., & Bianchi, C. N. (2009). The crab that came in from the cold: First record of *Paralithodes camtschaticus* (Tilesius, 1815) in the Mediterranean Sea. *Aquatic Invasions, 4*(4), 715–718. https://doi.org/10.3391/ai.2009.4.4.22

FAO. (2012). *Leptocybe invasa (Forest Pest Species Series)*. Food and Agriculture Organization of the United Nations. https://www.fao.org/forestry/13569-05912e0e2fe9054c3ed4904ae597e3310.pdf

FAO. (2020). *The State of the World Fisheries and Aquaculture 2020. Sustainability in action* (p. 224). Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/ca9229en

FAO. (2021). *FAO Fisheries and Aquaculture Division—Database on Introductions of Aquatic Species (DIAS)*. Fisheries and Aquaculture Division, Food and Agriculture Organization of the United Nations. https://www.fao.org/fishery/en/introsp/search

Faria, L. C., & Kitahara, M. V. (2020). Invasive corals hitchhiking in the Southwestern Atlantic. *Ecology, 101*(8), e03066. https://doi.org/10.1002/ecy.3066

Farrell, M. J., & Davies, T. J. (2019). Disease mortality in domesticated animals is predicted by host evolutionary relationships. *Proceedings of the National Academy of Sciences, 116*(16), Article 16. https://doi.org/10.1073/pnas.1817323116

Faulkner, K. T., Burness, A., Byrne, M. J., Kumschick, S., Peters, K., Robertson, M. P., Saccaggi, D. L., Weyl, O. L. F., & Williams, V. L. (2020). South Africa's pathways of introduction and dispersal and how they have changed over time. In B. W. van Wilgen, J. Measey, D. W. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological invasions in South Africa* (Vol. 14, pp. 311–352). Springer. https://doi.org/10.1007/978-3-030-32394-3_12

Faulkner, K. T., Robertson, M. P., Rouget, M., & Wilson, J. R. U. (2016). Understanding and managing the introduction pathways of alien taxa: South Africa as a case study. *Biological Invasions, 18*(1), Article 1. https://doi.org/10.1007/s10530-015-0990-4

Faulkner, K. T., Spear, D., Robertson, M. P., Rouget, M., & Wilson, J. R. U. (2015). An assessment of the information content of South African alien species databases. *Bothalia, 45*(1). https://doi.org/10.4102/abc.v45i1.1103

Feldmeier, S., Schefczyk, L., Wagner, N., Heinemann, G., Veith, M., & Lötters, S. (2016). Exploring the Distribution of the Spreading Lethal Salamander Chytrid Fungus in Its Invasive Range in Europe – A Macroecological Approach. *PLoS ONE, 11*(10), Article 10. https://doi.org/10.1371/journal.pone.0165682

Fenoglio, S., Bonada, N., Guareschi, S., López-Rodríguez, M. J., Millán, A., & Tierno de Figueroa, J. M. (2016). Freshwater ecosystems and aquatic insects: A paradox in biological invasions. *Biology Letters, 12*(4), Article 4. https://doi.org/10.1098/rsbl.2015.1075

Fernández de Alaiza García Madrigal, R., da Silva, U. de A. T., Tavares, C. P. dos S., & Ballester, E. L. C. (2018). Use of native and non-native shrimp (Penaeidae, Dendrobranchiata) in world shrimp farming. *Reviews in Aquaculture, 10*(4), 899–912. https://doi.org/10.1111/raq.12206

Fernández-Llamazares, Á., Lepofsky, D., Lertzman, K., Armstrong, C. G., Brondizio, E. S., Gavin, M. C., Lyver, P. O., Nicholas, G. P., Pascua, P., Reo, N. J., Reyes-García, V., Turner, N. J., Yletyinen, J., Anderson, E. N., Balée, W., Cariño, J., David-Chavez, D. M., Dunn, C. P., Garnett, S. C., … Vaughan, M. B. (2021). Scientists' Warning to Humanity on Threats to Indigenous and Local Knowledge Systems. *Journal of Ethnobiology, 41*(2). https://doi.org/10.2993/0278-0771-41.2.144

Ferreira, A. A., Welch, J. R., Santos, R. V., Gugelmin, S. A., Ea, C., & Coimbra Jr., C. E. A. (2012). Nutritional status and growth of indigenous Xavante children, Central Brazil. *Nutririon Journal, 11*. https://doi.org/10.1186/1475-2891-11-3

Ficetola, G. F., Maiorano, L., Falcucci, A., Dendoncker, N., Boitani, L., Padoa-Schioppa, E., Miaud, C., & Thuiller, W. (2010). Knowing the past to predict the future: Land-use change and the distribution of invasive bullfrogs. *Global Change Biology, 16*(2), 528–537. https://doi.org/10.1111/j.1365-2486.2009.01957.x

Figueroa López, N., & Brante, A. (2020). Estado actual del conocimiento de las bioinvasiones marinas en Venezuela: Temáticas desarrolladas y tendencia temporal. *Gayana (Concepción), 84*(1), 1–15. https://doi.org/10.4067/S0717-65382020000100001

Filz, K. J., Bohr, A., & Lötters, S. (2018). Abandoned Foreigners: Is the stage set for exotic pet reptiles to invade Central Europe? *Biodiversity and Conservation, 27*, 417–435. https://doi.org/10.1007/s10531-017-1444-3

Fine, P. V. A. (2002). The invasibility of tropical forests by exotic plants. *Journal of Tropical Ecology, 18*(5), 687–705. https://doi.org/10.1017/S0266467402002456

Fischer, M. L., Costa, L. C. M., & Nering, I. S. (2012). Utilização de recursos alimentares presentes no ambiente antrópico pelo caramujo gigante africano *Achatina fulica* Bowdich, 1822: Subsídios para o manejo. *Títulos não-correntes, 22*(2), 91–100. https://seer.sis.puc-campinas.edu.br/bioikos/article/view/736

Fisher, M. C., Garner, T. W. J., & Walker, S. F. (2009). Global Emergence of *Batrachochytrium dendrobatidis* and Amphibian Chytridiomycosis in Space, Time, and Host. *Annual Review of Microbiology, 63*(1), 291–310. https://doi.org/10.1146/annurev.micro.091208.073435

Fisher, M. C., Gurr, S. J., Cuomo, C. A., Blehert, D. S., Jin, H., Stukenbrock, E. H., Stajich, J. E., Kahmann, R., Boone, C., Denning, D. W., Gow, N. A. R., Klein, B. S., Kronstad, J. W., Sheppard, D. C., Taylor, J. W., Wright, G. D., Heitman, J., Casadevall, A., & Cowen, L. E. (2020). Threats Posed by the Fungal Kingdom to Humans, Wildlife, and Agriculture. *mBio, 11*(3), e00449-20. https://doi.org/10.1128/mBio.00449-20

Fisher, M. C., Henk, D. A., Briggs, C. J., Brownstein, J. S., Madoff, L. C., McCraw, S. L., & Gurr, S. J. (2012). Emerging fungal threats to animal, plant and ecosystem health. *Nature, 484*(7393), 186–194. https://doi.org/10.1038/nature10947

Floerl, O., & Inglis, G. J. (2005). Starting the invasion pathway: The interaction between source populations and human transport vectors. *Biological Invasions, 7*(4), 589–606. https://doi.org/10.1007/s10530-004-0952-8

Florencio, M., Lobo, J. M., & Bini, L. M. (2019). Biases in global effects of exotic species on local invertebrates: A systematic review. *Biological Invasions, 21*(10), 3043–3061. https://doi.org/10.1007/s10530-019-02062-1

Flory, S. L., & Clay, K. (2009). Effects of roads and forest successional age on experimental plant invasions. *Biological Conservation, 142*(11), 2531–2537. https://doi.org/10.1016/j.biocon.2009.05.024

Fones, H. N., Bebber, D. P., Chaloner, T. M., Kay, W. T., Steinberg, G., & Gurr, S. J. (2020). Threats to global food security from emerging fungal and oomycete crop pathogens. *Nature Food, 1*(6), 332–342. https://doi.org/10.1038/s43016-020-0075-0

Fonseca, C. R., Guadagnin, D. L., Emer, C., Masciadri, S., Germain, P., & Zalba, S. M. (2013). Invasive alien plants in the Pampas grasslands: A tri-national cooperation challenge. *Biological Invasions, 15*(8), 1751–1763. https://doi.org/10.1007/s10530-013-0406-2

Fonseca, É., Both, C., & Cechin, S. Z. (2019). Introduction pathways and socio-economic variables drive the distribution of alien amphibians and reptiles in a megadiverse country. *Diversity and Distributions, 25*(7), 1130–1141. https://doi.org/10.1111/ddi.12920

Forbes, B. C., & Jefferies, R. L. (1999). Revegetation of disturbed arctic sites: Constraints and applications. *Biological Conservation, 88*(1), 15–24. https://doi.org/10.1016/S0006-3207(98)00095-0

Forest Peoples Programme, International Indigenous Forum on Biodiversity, Indigenous Women's Biodiversity Network, Centres of Distinction on Indigenous and Local Knowledge, & Secretariat of the CBD. (2020). *Local Biodiversity Outlooks 2: The contributions of indigenous peoples and local communities to the implementation of the Strategic Plan for Biodiversity 2011–2020 and to renewing nature and cultures. A complement to the fifth edition of Global Biodiversity Outlook.* Moreton-in-Marsh, England. https://www.cbd.int/gbo/gbo5/publication/lbo-2-en.pdf

Forey, E., Lodhar, S. Y. F., Gopaul, S., Boehmer, H. J., & Chauvat, M. (2021). A functional trait-based approach to assess the impact of an alien palm invasion on plant and soil communities on a South Pacific island. *Austral Ecology, 46*(3), 398–410. https://doi.org/10.1111/AEC.12995

Forrest, B., Hopkins, G., Webb, S., & Tremblay, L. (2011). *Overview of Marine Biosecurity Risks from Finfish Aquaculture Development in the Waikato Region* (Cawthron Report No. 1871; p. 78). Waikato Regional Council. https://www.waikatoregion.govt.nz/assets/WRC/WRC-2019/TR201122.pdf

Forsyth, D. M., Ramsey, D. S. L., Perry, M., McKay, M., & Wright, E. F. (2018). Control history, longitude and multiple abiotic and biotic variables predict the abundances of invasive brushtail possums in New Zealand forests. *Biological Invasions, 20*(8), 2209–2225. https://doi.org/10.1007/s10530-018-1697-0

Forti, L. R., Becker, C. G., Tacioli, L., Pereira, V. R., Santos, A. C. F. A., Oliveira, I., Haddad, C. F. B., & Toledo, L. F. (2017). Perspectives on invasive amphibians in Brazil. *PLoS ONE, 12*(9), e0184703. https://doi.org/10.1371/journal.pone.0184703

Fournier, A., Penone, C., Grazia Pennino, M., & Courchamp, F. (2019). Predicting future invaders and future invasions. *Proceedings of the National Academy of Sciences of the United States of America, 116*(16), 7905–7910. https://doi.org/10.1073/pnas.1803456116

Fowler, A. E., Blakeslee, A. M. H., Canning-Clode, J., Repetto, M. F., Phillip, A. M., Carlton, J. T., Moser, F. C., Ruiz, G. M., & Miller, A. W. (2016). Opening Pandora's bait box: A potent vector for biological invasions of live marine species. *Diversity and Distributions, 22*(1), 30–42. https://doi.org/10.1111/ddi.12376

Foxcroft, L. C., Jarošík, V., Pyšek, P., Richardson, D. M., & Rouget, M. (2010). Protected-Area Boundaries as Filters of Plant Invasions: Park Boundaries and Invasive Plants. *Conservation Biology, 25*(2), 400–405. https://doi.org/10.1111/j.1523-1739.2010.01617.x

Foxcroft, L. C., Pyšek, P., Richardson, D. M., & Genovesi, P. (Eds.). (2013). Plant Invasions in *Protected Areas: Patterns, Problems and Challenges* (Vol. 7). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7

Foxcroft, L. C., Pyšek, P., Richardson, D. M., Genovesi, P., & MacFadyen, S. (2017). Plant invasion science in protected areas: Progress and priorities. *Biological Invasions, 19*(5), 1353–1378. https://doi.org/10.1007/s10530-016-1367-z

Foxcroft, L. C., Richardson, D. M., Rejmánek, M., & Pyšek, P. (2010). Alien plant invasions in tropical and sub-tropical savannas: Patterns, processes and prospects. *Biological Invasions, 12*(12), 3913–3933. https://doi.org/10.1007/s10530-010-9823-7

Foxcroft, L. C., Richardson, D. M., & Wilson, J. R. U. (2008). Ornamental Plants as Invasive Aliens: Problems and Solutions in Kruger National Park, South Africa. *Environmental Management, 41*(1), 32–51. https://doi.org/10.1007/s00267-007-9027-9

Foxcroft, L. C., Rouget, M., Richardson, D. M., & Macfadyen, S. M. (2004). Reconstructing 50 years of *Opuntia stricta* invasion in the Kruger National Park, South Africa: Environmental determinants and propagule pressure. *Diversity and Distributions, 10*(5–6), 427–437. https://doi.org/10.1111/j.1366-9516.2004.00117.x

Franco, A. C. S., Petry, A. C., Tavares, M. R., Guimarães, T. F. R., & dos Santos, L. N. (2022). Global distribution of the South American peacock basses *Cichla* spp. Follows human interference. *Fish and Fisheries, 23*(2), 407–421. https://doi.org/10.1111/faf.12624

Francour, P., Mangialajo, L., & Pastor, J. (2010). Mediterranean marine protected areas and non-indigenous fish spreading. In D. Golani & B. Appelbaum-Golani (Eds.), *Fish Invasions of the Mediterranean Sea: Change and Renewal* (pp. 127–144). Pensoft Publishers.

Franz, N. M., & Sterner, B. W. (2018). To increase trust, change the social design behind aggregated biodiversity data. *Database, 2018*(2018), bax100. https://doi.org/10.1093/database/bax100

Freestone, A. L., Osman, R. W., Ruiz, G. M., & Torchin, M. E. (2011). Stronger predation in the tropics shapes species richness patterns in marine communities. *Ecology, 92*(4), 983–993. https://doi.org/10.1890/09-2379.1

Frehse, F. de A., Braga, R. R., Nocera, G. A., & Vitule, J. R. S. (2016). Non-native species and invasion biology in a megadiverse country: Scientometric analysis and ecological interactions in Brazil. *Biological Invasions, 18*(12), 3713–3725. https://doi.org/10.1007/s10530-016-1260-9

Frelich, L. E., Blossey, B., Cameron, E. K., Dávalos, A., Eisenhauer, N., Fahey, T., Ferlian, O., Groffman, P. M., Larson, E., Loss, S. R., Maerz, J. C., Nuzzo, V., Yoo, K., & Reich, P. B. (2019). Side-swiped: Ecological cascades emanating from earthworm invasions. *Frontiers in Ecology and the Environment, 17*(9), 502–510. https://doi.org/10.1002/fee.2099

Frenot, Y., Chown, S. L., Whinam, J., Selkirk, P. M., Convey, P., Skotnicki, M., & Bergstrom, D. M. (2005). Biological invasions in the Antarctic: Extent, impacts and implications. *Biological Reviews, 80*(1), 45–72. https://doi.org/10.1017/S1464793104006542

Fridley, J. D. (2008). Of Asian Forests and European Fields: Eastern U.S. Plant Invasions in a Global Floristic Context. *PLoS ONE, 3*(11), e3630. https://doi.org/10.1371/journal.pone.0003630

Fridley, J. D. (2013). Plant invasions across the Northern Hemisphere: A deep-time perspective. *Annals of the New York Academy of Sciences, 1293*, 8–17. https://doi.org/10.1111/nyas.12107

Friedel, M. H. (2020). Unwelcome guests: A selective history of weed introductions to arid and semi-arid Australia. *Australian Journal of Botany, 68*(2), 75–99. https://doi.org/10.1071/BT20030

Fritts, T. H., & Leasman-Tanner, D. (2001). The Brown Tree Snake on Guam: How the Arrival of One Invasive Species Damaged the Ecology, Commerce, Electrical Systems and Human Health on Guam: A Comprehensive Information Source. In *The Brown Tree Snake on Guam: How the Arrival of One Invasive Species Damaged the Ecology, Commerce, Electrical Systems and Human Health on Guam: A Comprehensive Information Source* (Federal Government Series 2002–0009; Information and Technology Report, Vols. 2002–0009, p. 10). U.S. Fish and Wildlife Service. http://pubs.er.usgs.gov/publication/53889

Fritts, T. H., & Rodda, G. H. (1998). The role of introduced species in the degradation of island ecosystems: A case history of Guam. *Annual Review of Ecology and Systematics, 29*(1), 113–140. https://doi.org/10.1146/annurev.ecolsys.29.1.113

Froese, R., & Pauly, D. (Eds). (2015). *FishBase.* World Wide Web Electronic Publication. https://www.fishbase.se/

Fry, W. (2008). *Phytophthora infestans*: The plant (and *R* gene) destroyer. *Molecular Plant Pathology, 9*(3), 385–402. https://doi.org/10.1111/j.1364-3703.2007.00465.x

Fuentes, N., Marticorena, A., Saldaña, A., Jerez, V., Ortiz, J. C., Victoriano, P., Moreno, R. A., Larraín, J., Villaseñor-Parada, C., Palfner, G., Sánchez, P., & Pauchard, A. (2020). Multi-taxa inventory of naturalized species in Chile. *NeoBiota, 60*, 25–41. https://doi.org/10.3897/neobiota.60.55366

Fuentes, N., Ugarte, E., Kühn, I., & Klotz, S. (2008). Alien plants in Chile: Inferring invasion periods from herbarium records. *Biological Invasions, 10*(5), 649–657. https://doi.org/10.1007/s10530-007-9159-0

Fuentes-Lillo, E., Cuba-Díaz, M., Troncoso-Castro, J. M., & Rondanelli-Reyes, M. (2017). Seeds of non-native species in King George Island soil. *Antarctic Science, 29*(4), Article 4. https://doi.org/10.1017/S0954102017000037

Fuentes-Lillo, E., Lembrechts, J. J., Cavieres, L. A., Jiménez, A., Haider, S., Barros, A., & Pauchard, A. (2021). Anthropogenic factors overrule local abiotic variables in determining non-native plant invasions in mountains. *Biological Invasions, 23*(12), 3671–3686. https://doi.org/10.1007/s10530-021-02602-8

Fuller, P. L., Nico, L. G., & Williams, J. D. (1999). *Nonindigenous Fishes Introduced into Inland Waters of the United States.* American Fisheries Society. https://doi.org/10.47886/9781888569148

Gaertner, M., Larson, B. M. H., Irlich, U. M., Holmes, P. M., Stafford, L., van Wilgen, B. W., & Richardson, D. M. (2016). Managing invasive species in cities: A framework from Cape Town, South Africa. *Landscape and Urban Planning, 151*, 1–9. https://doi.org/10.1016/j.landurbplan.2016.03.010

Gaertner, M., Wilson, J. R. U., Cadotte, M. W., MacIvor, J. S., Zenni, R. D., & Richardson, D. M. (2017). Non-native species in urban environments: Patterns, processes, impacts and challenges. *Biological Invasions, 19*(12), 3461–3469. https://doi.org/10.1007/s10530-017-1598-7

Galasso, G., Conti, F., Peruzzi, L., Ardenghi, N. M. G., Banfi, E., Celesti-Grapow, L., Albano, A., Alessandrini, A., Bacchetta, G., Ballelli, S., Bandini Mazzanti, M., Barberis, G., Bernardo, L., Blasi, C., Bouvet, D., Bovio, M., Cecchi, L., Del Guacchio, E., Domina, G., … Bartolucci, F. (2018). An updated checklist of the vascular flora alien to Italy. *Plant Biosystems, 152*(3), 556–592. https://doi.org/10.1080/11263504.2018.1441197

Galera, H., Chwedorzewska, K. J., Korczak-Abshire, M., & Wódkiewicz, M. (2018). What affects the probability of biological invasions in Antarctica? Using an expanded conceptual framework to anticipate the risk of alien species expansion. *Biodiversity and Conservation, 27*(8), 1789–1809. https://doi.org/10.1007/s10531-018-1547-5

Galil, B. S. (2000). A sea under siege – alien species in the Mediterranean. *Biological Invasions, 2*(2), 177–186. https://doi.org/10.1023/A:1010057010476

Galil, B. S. (2011). The Alien Crustaceans in the Mediterranean Sea: An Historical Review. In B. S. Galil, P. Clark, & J. Carlton (Eds.), *In the Wrong Place—Alien Marine Crustaceans: Distribution, Biology and Impacts* (Vol. 6, pp. 377–401). Springer. https://doi.org/10.1007/978-94-007-0591-3_13

Galil, B. S. (2017). Eyes Wide Shut: Managing Bio-Invasions in Mediterranean Marine Protected Areas. In P. D. Goriup (Ed.), *Management of Marine Protected Areas* (pp. 187–206). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781119075806.ch10

Galil, B. S., Boero, F., Campbell, M. L., Carlton, J. T., Cook, E., Fraschetti, S., Gollasch, S., Hewitt, C. L., Jelmert, A., Macpherson, E., Marchini, A., McKenzie, C., Minchin, D., Occhipinti-Ambrogi, A., Ojaveer, H., Olenin, S., Piraino, S., & Ruiz, G. M. (2015). 'Double trouble': The expansion of the Suez Canal and marine bioinvasions in the Mediterranean Sea. *Biological Invasions, 17*(4), 973–976. https://doi.org/10.1007/s10530-014-0778-y

Galil, B. S., Boero, F., Fraschetti, S., Piraino, S., Campbell, M., Hewitt, C., Carlton, J. T., Cook, E., Jelmert, A., Macpherson, E., Marchini, A., Occhipinti-Ambrogi, A., Mckenzie, C., Minchin, D., Ojaveer, H., Olenin, S., & Ruiz, G. (2015). The Enlargement of the Suez Canal and Introduction of Non-Indigenous Species to the Mediterranean Sea. *Limnology and Oceanography Bulletin, 24*(2), 43–45. https://doi.org/10.1002/lob.10036

Galil, B. S., Clark, P. F., & Carlton, J. T. (Eds.). (2011). *In the Wrong Place: Alien Marine Crustaceans: Distribution, Biology and Impacts* (Vol. 6). Springer Science & Business Media. https://doi.org/10.1007/978-94-007-0591-3

Galil, B. S., Danovaro, R., Rothman, S. B. S., Gevili, R., & Goren, M. (2019). Invasive biota in the deep-sea Mediterranean: An emerging issue in marine conservation and management. *Biological Invasions, 21*(2), 281–288. https://doi.org/10.1007/s10530-018-1826-9

Galil, B. S., Marchini, A., Occhipinti-Ambrogi, A., Minchin, D., Narščius, A., Ojaveer, H., & Olenin, S. (2014). International arrivals: Widespread bioinvasions in European Seas. *Ethology Ecology & Evolution, 26*(2–3), 152–171. https://doi.org/10.1080/03949370.2014.897651

Galil, B. S., Marchini, A., Occhipinti-Ambrogi, A., & Ojaveer, H. (2017). The enlargement of the Suez Canal – Erythraean introductions and management challenges. *Management of Biological Invasions, 8*(2), 141–152. https://doi.org/10.3391/mbi.2017.8.2.02

Galil, B. S., Mienis, H. K., Hoffman, R., & Goren, M. (2021a). Correction to: Non-indigenous species along the Israeli Mediterranean coast: tally, policy, outlook. *Hydrobiologia, 848*(9), 2031. https://doi.org/10.1007/s10750-020-04502-9

Galil, B. S., Mienis, H. K., Hoffman, R., & Goren, M. (2021b). Non-indigenous species along the Israeli Mediterranean coast: Tally, policy, outlook. *Hydrobiologia, 848*(9), 2011–2029. https://doi.org/10.1007/s10750-020-04420-w

Galil, B. S., Nehring, S., & Panov, V. (2007). Waterways as Invasion Highways – Impact of Climate Change and Globalization. In W. Nentwig (Ed.), *Biological Invasions* (Vol. 193, pp. 59–74). Springer Berlin Heidelberg.

Gallardo, B., Aldridge, D. C., González-Moreno, P., Pergl, J., Pizarro, M., Pyšek, P., Thuiller, W., Yesson, C., & Vilà, M. (2017). Protected areas offer refuge from invasive species spreading under climate change. *Global Change Biology, 23*(12), 5331–5343. https://doi.org/10.1111/gcb.13798

García, N., Cuttelod, A., & Abdul Malak, D. (2010). *The Status and Distribution of Freshwater Biodiversity in Northern Africa.* (p. xiii+141). IUCN. https://portals.iucn.org/library/sites/library/files/documents/RL-61-002.pdf

García-Díaz, P., & Cassey, P. (2014). Patterns of transport and introduction of exotic amphibians in Australia. *Diversity and Distributions, 20*, 455–466. https://doi.org/10.1111/ddi.12176

García-Díaz, P., Ramsey, D. S. L., Woolnough, A. P., Franch, M., Llorente, G. A., Montori, A., Buenetxea, X., Larrinaga, A. R., Lasceve, M., Álvarez, A., Traverso, J. M., Valdeón, A., Crespo, A., Rada, V., Ayllón, E., Sancho, V., Lacomba, J. I., Bataller, J. V., & Lizana, M. (2017). Challenges in confirming eradication success of invasive red-eared sliders. *Biological Invasions, 19*(9), 2739–2750. https://doi.org/10.1007/s10530-017-1480-7

García-Díaz, P., Ross, J. V., Ayres, C., & Cassey, P. (2015). Understanding the biological invasion risk posed by the global wildlife trade: Propagule pressure drives the introduction and establishment of Nearctic turtles. *Global Change Biology, 21*(3), 1078–1091. https://doi.org/10.1111/gcb.12790

Garnett, S. T., Burgess, N. D., Fa, J. E., Fernández-Llamazares, Á., Molnár, Z., Robinson, C. J., Watson, J. E. M., Zander,

K. K., Austin, B., Brondizio, E. S., Collier, N. F., Duncan, T., Ellis, E., Geyle, H., Jackson, M. V., Jonas, H., Malmer, P., McGowan, B., Sivongxay, A., & Leiper, I. (2018). A spatial overview of the global importance of Indigenous lands for conservation. *Nature Sustainability, 1*(7), 369–374. https://doi.org/10.1038/s41893-018-0100-6

Gassó, N., Pyšek, P., Vilà, M., & Williamson, M. (2010). Spreading to a limit: The time required for a neophyte to reach its maximum range. *Diversity and Distributions, 16*(2), 310–311. https://doi.org/10.1111/j.1472-4642.2010.00647.x

Gaston, K. J., Jackson, S. F., Cantú-Salazar, L., & Cruz-Piñón, G. (2008). The Ecological Performance of Protected Areas. *Annual Review of Ecology, Evolution, and Systematics, 39*(1), 93–113. https://doi.org/10.1146/annurev.ecolsys.39.110707.173529

Gautam, M. R., Chief, K., & Smith, W. J. (2013). Climate change in arid lands and Native American socioeconomic vulnerability: The case of the Pyramid Lake Paiute Tribe. *Climatic Change, 120*(3), 585–599. https://doi.org/10.1007/s10584-013-0737-0

Genovesi, P., Bacher, S., Kobelt, M., Pascal, M., & Scalera, R. (2009). Alien Mammals of Europe. In *Handbook of Alien Species in Europe* (Vol. 3, pp. 119–128). Springer Netherlands. https://doi.org/10.1007/978-1-4020-8280-1_9

Genovesi, P., Carnevali, L., Alonzi, A., & Scalera, R. (2012). Alien mammals in Europe: Updated numbers and trends, and assessment of the effects on biodiversity. *Integrative Zoology, 7*(3), 247–253. https://doi.org/10.1111/j.1749-4877.2012.00309.x

Géron, C., Lembrechts, J. J., Borgelt, J., Lenoir, J., Hamdi, R., Mahy, G., Nijs, I., & Monty, A. (2021). Urban alien plants in temperate oceanic regions of Europe originate from warmer native ranges. *Biological Invasions, 23*(6), 1765–1779. https://doi.org/10.1007/s10530-021-02469-9

Gessler, C., Pertot, I., & Perazzolli, M. (2011). *Plasmopara viticola*: A review of knowledge on downy mildew of grapevine and effective disease management. *Phytopathologia Mediterranea, 50*(1), 3–44. JSTOR. http://www.jstor.org/stable/26458675

Gettys, L. A. (2019). Breaking Bad: Native Aquatic Plants Gone Rogue and the Invasive Species That Inspire Them. *HortTechnology, 29*(5), 559–566. https://doi.org/10.21273/HORTTECH04333-19

Gherardi, F. (2010). Invasive crayfish and freshwater fishes of the world. *Revue Scientifique Et Technique (International Office of Epizootics), 29*(2), 241–254. https://doi.org/10.20506/rst.29.2.1973

Gherardi, F., Aquiloni, L., Diéguez-Uribeondo, J., & Tricarico, E. (2011). Managing invasive crayfish: Is there a hope? *Aquatic Sciences, 73*(2), 185–200. https://doi.org/10.1007/s00027-011-0181-z

Gherardi, F., Britton, J. R., Mavuti, K. M., Pacini, N., Grey, J., Tricarico, E., & Harper, D. M. (2011). A review of allodiversity in Lake Naivasha, Kenya: Developing conservation actions to protect East African lakes from the negative impacts of alien species. *Biological Conservation, 144*(11), 2585–2596. https://doi.org/10.1016/j.biocon.2011.07.020

Gherardi, F., Gollasch, S., Minchin, D., Olenin, S., & Panov, V. (2009). Alien Invertebrates and Fish in European Inland Waters. In *Handbook of Alien Species in Europe* (Vol. 3, pp. 81–92). Springer. https://doi.org/10.1007/978-1-4020-8280-1_6

Giakoumi, S., Pey, A., Di Franco, A., Francour, P., Kizilkaya, Z., Arda, Y., Raybaud, V., & Guidetti, P. (2019). Exploring the relationships between marine protected areas and invasive fish in the world's most invaded sea. *Ecological Applications, 29*(1), e01809. https://doi.org/10.1002/eap.1809

Gilardi, G., Demarchi, S., Garibaldi, A., & Gullino, M. L. (2013). Management of downy mildew of sweet basil *(Ocimum basilicum)* caused by *Peronospora belbahrii* by means of resistance inducers, fungicides, biocontrol agents and natural products. *Phytoparasitica, 41*(1), 59–72. https://doi.org/10.1007/s12600-012-0264-y

Giovas, C. M., LeFebvre, M. J., & Fitzpatrick, S. M. (2012). New records for prehistoric introduction of Neotropical mammals to the West Indies: Evidence from Carriacou, Lesser Antilles: Neotropical mammal translocation to Carriacou. *Journal of Biogeography, 39*(3), 476–487. https://doi.org/10.1111/j.1365-2699.2011.02630.x

Gippet, J. M. W., & Bertelsmeier, C. (2021). Invasiveness is linked to greater commercial success in the global pet trade. *Proceedings of the National Academy of Sciences, 118*(14), e2016337118. https://doi.org/10.1073/pnas.2016337118

Gippet, J. M. W., Liebhold, A. M., Fenn-Moltu, G., & Bertelsmeier, C. (2019). Human-mediated dispersal in insects. *Current Opinion in Insect Science, 35*, 96–102. https://doi.org/10.1016/j.cois.2019.07.005

Giraud, T., Pedersen, J. S., & Keller, L. (2002). Evolution of supercolonies: The Argentine ants of southern Europe. *Proceedings of the National Academy of Sciences, 99*(9), 6075–6079. https://doi.org/10.1073/pnas.092694199

GISD. (2019). Species profile: *Suncus murinus*. In *Global Invasive Species Database.* http://www.iucngisd.org/gisd/speciesname/Suncus+murinus

Global Islands Network. (2021). *Global Island Database.* GIN – Global Islands Network – the Number One Source of Information on Islands and Island Issues. http://www.globalislands.net/about/gid_functions.php

Godde, P. M., Price, M. F., & Zimmermann, F. M. (Eds.). (2000). *Tourism and Development in Mountain Regions.* CABI.

Godoy, O., Saldaña, A., Fuentes, N., Valladares, F., & Gianoli, E. (2011). Forests are not immune to plant invasions: Phenotypic plasticity and local adaptation allow *Prunella vulgaris* to colonize a temperate evergreen rainforest. *Biological Invasions, 13*(7), 1615–1625. https://doi.org/10.1007/s10530-010-9919-0

Goldsmit, J., McKindsey, C. W., Schlegel, R. W., Stewart, D. B., Archambault, P., & Howland, K. L. (2020). What and where? Predicting invasion hotspots in the Arctic marine realm. *Global Change Biology, 26*(9), 4752–4771. https://doi.org/10.1111/gcb.15159

Goldson, S. L. (2011). Biosecurity, risk and policy: A New Zealand perspective. *Journal Für Verbraucherschutz Und Lebensmittelsicherheit, 6*(Suppl 1), 41–47. https://doi.org/10.1007/s00003-011-0673-8

Gollasch, S. (2006). Overview on introduced aquatic species in European navigational and adjacent waters. *Helgoland Marine Research, 60*, 84–89. https://doi.org/10.1007/s10152-006-0022-y

Gollasch, S., & Nehring, S. (2006). National checklist for aquatic alien species in Germany. *Aquatic Invasions, 1*(4), 245–269. https://doi.org/10.3391/ai.2006.1.4.8

González, A., & Santelices, B. (2004). A dichotomous species of *Codium* (Bryopsidales, Chlorophyta) is colonizing northern Chile. *Revista Chilena de Historia Natural, 77*(2), 293–304. https://doi.org/10.4067/S0716-078X2004000200008

González-Sánchez, V. H., Johnson, J. D., González-Solís, D., Fucsko, L. A., & Wilson, L. D. (2021). A review of the introduced herpetofauna of Mexico and Central America, with comments on the effects of invasive species and biosecurity methodology. *ZooKeys, 1022*, 79–154. https://doi.org/10.3897/zookeys.1022.51422

Goodman, P. S. (2003). Assessing Management Effectiveness and Setting Priorities in Protected Areas in KwaZulu-Natal. *BioScience, 53*(9), 843–850. https://doi.org/10.1641/0006-3568(2003)053[0843:AMEASP]2.0.CO;2

Goren, M., & Ortal, R. (1999). Biogeography, diversity and conservation of the inland water fish communities in Israel. *Biological Conservation, 89*(1), 1–9. https://doi.org/10.1016/S0006-3207(98)00127-X

Görg, M., Ploch, S., Kruse, J., Kummer, V., Runge, F., Choi, Y.-J., & Thines, M. (2017). Revision of *Plasmopara* (Oomycota, Peronosporales) parasitic to *Impatiens. Mycological Progress, 16*(8), 791–799. https://doi.org/10.1007/s11557-017-1316-y

Gorgone-Barbosa, E., Pivello, V. R., Bautista, S., Zupo, T., Rissi, M. N., & Fidelis, A. (2015). How can an invasive grass affect fire behavior in a tropical savanna? A community and individual plant level approach. *Biological Invasions, 17*(1), 423–431. https://doi.org/10.1007/s10530-014-0740-z

Gormley, A. M., Penelope Holland, E., Pech, R. P., Thomson, C., & Reddiex, B. (2012). Impacts of an invasive herbivore on indigenous forests. *Journal of Applied Ecology, 49*(6), 1296–1305. https://doi.org/10.1111/j.1365-2664.2012.02219.x

Goulson, D. (2003). Effects of Introduced Bees on Native Ecosystems. *Annual Review of Ecology, Evolution, and Systematics, 34*(1), 1–26. https://doi.org/10.1146/annurev.ecolsys.34.011802.132355

Government of India. (2005). *Ministry of Environment and Forests, Country report on status of forest invasive alien species in India* (p. 74). Asia-Pacific Forest Invasive Species Network. https://apfisn.net/wp-content/uploads/2018/07/India.pdf

Government of Myanmar. (2005). *Country report of status of forest invasive alien species in Myanmar* (p. 31). Asia-Pacific Forest Invasive Species Network. https://apfisn.net/wp-content/uploads/2018/07/Myanmar.pdf

Gozlan, R. E. (2008). Introduction of non-native freshwater fish: Is it all bad? *Fish and Fisheries, 9*(1), 106–115. https://doi.org/10.1111/j.1467-2979.2007.00267.x

Gracia, A., Medellin Mora, J., Gil Agudelo, D., & Puentes G., V. (2011). *Guía de las especies introducidas marinas y costeras de Colombia.* (Vol. 23). Instituto de Investigaciones Marinas y Costeras – INVEMAR y Ministerio de Ambiente y Desarrollo Sostenible. https://aquadocs.org/handle/1834/8275

Greve, M., Mathakutha, R., Steyn, C., & Chown, S. L. (2017). Terrestrial invasions on sub-Antarctic Marion and Prince Edward Islands. *Bothalia, 47*(2). https://doi.org/10.4102/abc.v47i2.2143

Griffiths, C. L., Hockey, P. A. R., Van Erkom Schurink, C., & Le Roux, P. J. (1992). Marine invasive aliens on South African shores: Implications for community structure and trophic functioning. *South African Journal of Marine Science, 12*(1), 713–722. https://doi.org/10.2989/02577619209504736

Griffiths, C. L., Robinson, T. B., & Mead, A. (2009). The Status and Distribution of Marine Alien Species in South Africa. In G. Rilov & J. A. Crooks (Eds.), *Biological Invasions in Marine Ecosystems* (Vol. 204, pp. 393–408). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-540-79236-9_23

Groom, Q. J., Adriaens, T., Desmet, P., Simpson, A., De Wever, A., Bazos, I., Cardoso, A. C., Charles, L., Christopoulou, A., Gazda, A., Helmisaari, H., Hobern, D., Josefsson, M., Lucy, F., Marisavljevic, D., Oszako, T., Pergl, J., Petrovic-Obradovic, O., Prévot, C., … Vanderhoeven, S. (2017). Seven Recommendations to Make Your Invasive Alien Species Data More Useful. *Frontiers in Applied Mathematics and Statistics, 3*, 13. https://doi.org/10.3389/fams.2017.00013

Groom, Q. J., Desmet, P., Reyserhove, L., Adriaens, T., Oldoni, D., Vanderhoeven, S., Baskauf, S. J., Chapman, A., McGeoch, M. A., Walls, R., Wieczorek, J., Wilson, J. R. U., Zermoglio, P. F., & Simpson, A. (2019). Improving Darwin Core for research and management of alien species. *Biodiversity Information Science and Standards, 3,* e38084. https://doi.org/10.3897/biss.3.38084

Gross, A., Petitcollin, C., Dutech, C., Ly, B., Massot, M., d'Arcier, J. F., Dubois, L., Saint-Jean, G., & Desprez-Loustau, M.-L. (2021). Hidden invasion and niche contraction revealed by herbaria specimens in the fungal complex causing oak powdery mildew in Europe. *Biological Invasions, 23*(3), 885–901. https://doi.org/10.1007/s10530-020-02409-z

Gruenwald, N. J. (2012). Novel insights into the emergence of pathogens: The case of chestnut blight. *Molecular Ecology, 21*(16), 3896–3897. https://doi.org/10.1111/j.1365-294X.2012.05597.x

Guan, H., Fan, J., & Li, Y. (2019). The impact of different introduced artificial grassland species combinations on community biomass and species diversity in temperate steppe of the Qinghai-Tibetan Plateau. *Acta Prataculturae Sinica, 28*(9), 192–201. https://doi.org/10.11686/cyxb2019100

Guareschi, S., Coccia, C., Sánchez-Fernández, D., Carbonell, J. A., Velasco, J., Boyero, L., Green, A. J., & Millán, A. (2013). How far could the alien boatman *Trichocorixa verticalis verticalis* spread? Worldwide estimation of its current and future potential distribution. *PLoS ONE, 8*(3), e59757. https://doi.org/10.1371/journal.pone.0059757

Guarnaccia, V., Gehrmann, T., Silva-Junior, G. J., Fourie, P. H., Haridas, S., Vu, D., Spatafora, J., Martin, F. M., Robert, V., Grigoriev, I. V., Groenewald, J. Z., & Crous, P. W. (2019). *Phyllosticta citricarpa* and sister species of global importance to *Citrus. Molecular Plant Pathology, 20*(12), 1619–1635. https://doi.org/10.1111/mpp.12861

Guidetti, P., Baiata, P., Ballesteros, E., Di Franco, A., Hereu, B., Macpherson, E., Micheli, F., Pais, A., Panzalis, P., Rosenberg, A. A., Zabala, M., & Sala, E. (2014). Large-scale assessment of mediterranean marine protected areas effects on fish assemblages. *PLoS ONE, 9*(4), e91841. https://doi.org/10.1371/journal.pone.0091841

Guillemin, M.-L., Faugeron, S., Destombe, C., Viard, F., Correa, J. A., & Valero, M. (2008). Genetic variation in wild and cultivated populations of the haploid-diploid red alga *Gracilaria chilensis*: How farming practices favor asexual reproduction and heterozygosity. *Evolution; International Journal of Organic Evolution, 62*(6), 1500–1519. https://doi.org/10.1111/j.1558-5646.2008.00373.x

Guiry, M. D., & Guiry, G. M. (2021). *AlgaeBase*. World-Wide Electronic Publication, National University of Ireland, Galway. https://www.algaebase.org/

Guo, Q., Cade, B. S., Dawson, W., Essl, F., Kreft, H., Pergl, J., van Kleunen, M., Weigelt, P., Winter, M., & Pyšek, P. (2021). Latitudinal patterns of alien plant invasions. *Journal of Biogeography, 48*(2), 253–262. https://doi.org/10.1111/jbi.13943

Guo, X. (2009). Use and exchange of genetic resources in molluscan aquaculture. *Reviews in Aquaculture, 1*(3-4), 251–259. https://doi.org/10.1111/j.1753-5131.2009.01014.x

Habit, E., González, J., Ortiz-Sandoval, J., Elgueta, A., & Sobenes, C. (2015). Effects of salmonid invasion in rivers and lakes of Chile. *Ecosistemas, 24*(1), 43–51. https://doi.org/10.7818/ECOS.2015.24-1.08

Habit, E., Piedra, P., Ruzzante, D. E., Walde, S. J., Belk, M. C., Cussac, V. E., Gonzalez, J., & Colin, N. (2010). Changes in the distribution of native fishes in response to introduced species and other anthropogenic effects. *Global Ecology and Biogeography, 19*(5), 697–710. https://doi.org/10.1111/j.1466-8238.2010.00541.x

Hack, W. H. (1949). Nota sobre un colémbolo de la Antartida Argentina Achorutes viaticus Tullberg. *Notas Del Museo de La Plata, 14*, 211–212.

Haider, S., Alexander, J., Dietz, H., Trepl, L., Edwards, P. J., & Kueffer, C. (2010). The role of bioclimatic origin, residence time and habitat context in shaping non-native plant distributions along an altitudinal gradient. *Biological Invasions, 12*(12), 4003–4018. https://doi.org/10.1007/s10530-010-9815-7

Haider, S., Kueffer, C., Bruelheide, H., Seipel, T., Alexander, J. M., Rew, L. J., Arévalo, J. R., Cavieres, L. A., McDougall, K. L., Milbau, A., Naylor, B. J., Speziale, K., & Pauchard, A. (2018). Mountain roads and non-native species modify elevational patterns of plant diversity. *Global Ecology and Biogeography, 27*(6), 667–678. https://doi.org/10.1111/geb.12727

Haider, S., Lembrechts, J., McDougall, K., Pauchard, A., Alexander, J. M., Barros, A., Cavieres, L., Rashid, I., Rew, L., Aleksanyan, A., Sierra, J. A., Aschero, V., Chisholm, C., Clark, V. R., Clavel, J., Daehler, C., Dar, P., Dietz, H., Dimarco, R., … Seipel, T. (2022). Think globally, measure locally: The MIREN standardized protocol for monitoring species distributions along elevation gradients. *Ecology and Evolution, 12*(2), e8590. https://doi.org/10.1002/ece3.8590

Hale, C. M., Frelich, L. E., & Reich, P. B. (2005). Exotic European earthworm invasion dynamics in northern hardwood forests of Minnesota, USA. *Ecological Applications, 15*(3), 848–860. https://doi.org/10.1890/03-5345

Hamilton, A. C., Karamura, D., & Kakudidi, E. (2016). History and conservation of wild and cultivated plant diversity in Uganda: Forest species and banana varieties as case studies. *Plant Diversity, 38*(1), 23–44. https://doi.org/10.1016/j.pld.2016.04.001

Handley, L. L. (2015). How will the 'molecular revolution' contribute to biological recording?: The Molecular Revolution in Biological Recording. *Biological Journal of the Linnean Society, 115*(3), 750–766. https://doi.org/10.1111/bij.12516

Hänfling, B., Edwards, F., & Gherardi, F. (2011). Invasive alien Crustacea: Dispersal, establishment, impact and control. *BioControl, 56*(4), 573–595. https://doi.org/10.1007/s10526-011-9380-8

Hao, T., Guillera-Arroita, G., May, T. W., Lahoz-Monfort, J. J., & Elith, J. (2020). Using Species Distribution Models For Fungi. *Fungal Biology Reviews, 34*(2), 74–88. https://doi.org/10.1016/j.fbr.2020.01.002

Haram, L. E., Carlton, J. T., Centurioni, L., Crowley, M., Hafner, J., Maximenko, N., Murray, C. C., Shcherbina, A. Y., Hormann, V., Wright, C., & Ruiz, G. M. (2021). Emergence of a neopelagic community through the establishment of coastal species on the high seas. *Nature Communications, 12*(1), 6885. https://doi.org/10.1038/s41467-021-27188-6

Harrison, Y. A., & Shackleton, C. M. (1999). Resilience of South African communal grazing lands after the removal of high grazing pressure. *Land Degradation & Development, 10*(3), 225–239. https://doi.org/10.1002/(SICI)1099-145X(199905/06)10:3<225::AID-LDR337>3.0.CO;2-T

Harvey, B., Soto, D., Carolsfeld, J., Beveridge, M., & Bartley, D. M. (2017). *Planning for aquaculture diversification: The importance of climate change and other drivers* (p. 166). FAO Fisheries and Aquaculture Proceedings No. 47. https://www.fao.org/3/i7358e/i7358e.pdf

Haslob, H., Clemmesen, C., Schaber, M., Hinrichsen, H. H., Schmidt, J. O., Voss, R., Kraus, G., & Köster, F. W. (2007). Invading *Mnemiopsis leidyi* as a potential threat to Baltic fish. *Marine Ecology – Progress Series, 349*, 303–306. https://doi.org/10.3354/meps07283

Haubrock, P. J., Oficialdegui, F. J., Zeng, Y., Patoka, J., Yeo, D. C. J., & Kouba, A. (2021). The redclaw crayfish: A prominent aquaculture species with invasive potential in tropical and subtropical biodiversity hotspots. *Reviews in Aquaculture, 13*(3), 1488–1530. https://doi.org/10.1111/raq.12531

Häussermann, V., & Försterra, G. (2001). A new species of sea anemone from Chile, *Anemonia alicemartinae* n. Sp. (Cnidaria: Anthozoa). An invader or an indicator for environmental change in shallow water? *Organisms Diversity & Evolution, 1*(3), 211–224. https://doi.org/10.1078/1439-6092-00018

Hayes, K. R., Inglis, G. J., & Barry, S. C. (2019). The Assessment and Management of Marine Pest Risks Posed by Shipping: The Australian and New Zealand Experience. *Frontiers in Marine Science, 6*, 489. https://doi.org/10.3389/fmars.2019.00489

Hays, W. S. T., & Conant, Sheila. (2007). Biology and Impacts of Pacific Island Invasive Species. 1. A Worldwide Review of Effects of the Small Indian Mongoose, *Herpestes javanicus* (Carnivora: Herpestidae). *Pacific Science, 61*(1), 3–16. https://doi.org/10.1353/psc.2007.0006

He, S., Worner, S. P., & Ikeda, T. (2012). Modeling the potential global distribution of light brown apple moth *Epiphyas postvittana* (Lepidoptera: Tortricidae) using CLIMEX. *Journal of Asia-Pacific Entomology, 15*(3, SI), 479–485. https://doi.org/10.1016/j.aspen.2012.01.004

Heberling, J. M., Jo, I., Kozhevnikov, A., Lee, H., & Fridley, J. D. (2017). Biotic interchange in the Anthropocene: Strong asymmetry in East Asian and eastern North American plant invasions. *Global Ecology and Biogeography, 26*(4), 447–458. https://doi.org/10.1111/geb.12551

Hebert, P. D. N., & Cristescu, M. E. A. (2002). Genetic perspectives on invasions: The case of the Cladocera. *Canadian Journal of Fisheries and Aquatic Sciences, 59*(7), 1229–1234. https://doi.org/10.1139/f02-091

Heinsohn, T. (2003). Animal translocation: Long-term human influences on the vertebrate zoogeography of Australasia (natural dispersal versus ethnophoresy). *Australian Zoologist, 32*(3), 351–376. https://doi.org/10.7882/AZ.2002.014

Hejda, M., Pyšek, P., Pergl, J., Sádlo, J., Chytrý, M., & Jarošík, V. (2009). Invasion success of alien plants: Do habitat affinities in the native distribution range matter? *Global Ecology and Biogeography, 18*(3), 372–382. https://doi.org/10.1111/j.1466-8238.2009.00445.x

Helmer, E. H., Ruzycki, T. S., Benner, J., Voggesser, S. M., Scobie, B. P., Park, C., Fanning, D. W., & Ramnarine, S. (2012). Detailed maps of tropical forest types are within reach: Forest tree communities for Trinidad and Tobago mapped with multiseason Landsat and multiseason fine-resolution imagery. *Forest Ecology and Management, 279*, 147–166. https://doi.org/10.1016/j.foreco.2012.05.016

Helmus, M. R., Mahler, D. L., & Losos, J. B. (2014). Island biogeography of the Anthropocene. *Nature, 513*(7519), 543–546. https://doi.org/10.1038/nature13739

Heluta, V., Takamatsu, S., Voytyuk, S., & Shiroya, Y. (2009). *Erysiphe kenjiana* (Erysiphales), a new invasive fungus in Europe. *Mycological Progress, 8*(4), 367–375. https://doi.org/10.1007/s11557-009-0610-8

Henderson, L. (1998). Invasive alien woody plants of the southern and southwestern Cape region, South Africa. *Bothalia, 28*(1), 91–112. https://doi.org/10.4102/abc.v28i1.624

Henderson, L. (2006). Comparisons of invasive plants in southern Africa originating from southern temperate, northern temperate and tropical regions. *Bothalia, 36*(2), 201–222. https://doi.org/10.4102/abc.v36i2.362

Henderson, L. (2007). Invasive, naturalized and casual alien plants in southern Africa: A summary based on the Southern African Plant Invaders Atlas (SAPIA). *Bothalia, 37*(2), 215–248. https://doi.org/10.4102/abc.v37i2.322

Henderson, L., & Wells, M. J. (1986). Alien plant invasions in the grassland and savanna biomes. In I. A. W. Macdonald, F. J. Kruger, & A. A. Ferrar (Eds.), *The ecology and management of biological invasions in southern Africa* (pp. 109–117). Oxford University Press. https://www.cabdirect.org/cabdirect/abstract/19880715517

Henderson, L., & Wilson, J. R. U. (2017). Changes in the composition and distribution of alien plants in South Africa: An update from the Southern African Plant Invaders Atlas. *Bothalia, 47*(2), a2172. https://doi.org/10.4102/abc.v47i2.2172

Henderson, S., Dawson, T. P., & Whittaker, R. J. (2006). Progress in invasive plants research. *Progress in Physical Geography: Earth and Environment, 30*(1), 25–46. https://doi.org/10.1191/0309133306pp468ra

Hendrix, P. F. (2006). Biological invasions belowground—Earthworms as invasive species. *Biological Invasions, 8*(6), 1201–1204. https://doi.org/10.1007/s10530-006-9048-y

Hendrix, P. F., & Bohlen, P. J. (2002). Exotic Earthworm Invasions in North America: Ecological and Policy Implications: Expanding global commerce may be increasing the likelihood of exotic earthworm invasions, which could have negative implications for soil processes, other animal and plant species, and importation of certain pathogens. *BioScience, 52*(9), 801–811. https://doi.org/10.1641/0006-3568(2002)052[0801:EEIINA]2.0.CO;2

Hendrix, P. F., Callaham, M. A., Drake, J. M., Huang, C.-Y., James, S. W., Snyder, B. A., & Zhang, W. (2008). Pandora's Box Contained Bait: The Global Problem of Introduced Earthworms. *Annual Review of Ecology, Evolution, and Systematics, 39*(1), 593–613. https://doi.org/10.1146/annurev.ecolsys.39.110707.173426

Hendry, A. P. (2020). *Eco-evolutionary Dynamics.* https://press.princeton.edu/books/paperback/9780691204178/eco-evolutionary-dynamics

Herbert, D. G. (2010). *The introduced terrestrial Mollusca of South Africa* (Vol. 15). South African National Biodiversity Institute.

Herder, J. E., Valentini, A., Bellemain, E., Dejean, T., van Delft, J. J. C. W., Thomsen, P. F., & Taberlet, P. (2014). *Environmental DNA – a review of the possible applications for the detection of (invasive) species* (Report 2013-104). Netherlands Food and Consumer Product Safety Authority. https://doi.org/10.13140/RG.2.1.4002.1208

Herms, D. A., & McCullough, D. G. (2014). Emerald Ash Borer Invasion of North America: History, Biology, Ecology, Impacts, and Management. *Annual Review of Entomology, 59*(1), 13–30. https://doi.org/10.1146/annurev-ento-011613-162051

Hernández, G., Lahmann, E. J., & Pérez-Gil Salcido, Ramón. (2002). *Invasores en Mesoamérica y el Caribe: Resultados del Taller sobre especies invasoras : ante los retos de su presencia en Mesoamérica y el Caribe.* UICN, Unión Mundial para la Naturaleza. https://portals.iucn.org/library/node/8032

Heshmati, I., Khorasani, N., Shams-Esfandabad, B., & Riazi, B. (2019). Forthcoming risk of *Prosopis juliflora* global invasion triggered by climate change: Implications for environmental monitoring and risk assessment. *Environmental Monitoring and Assessment, 191*(2), 72. https://doi.org/10.1007/s10661-018-7154-9

Hewitt, C. L., & Campbell, M. L. (2010). *The relative contribution of vectors to the introduction and translocation of invasive marine species.* The Department of Agriculture, Fisheries and Forestry (DAFF). https://www.marinepests.gov.au/sites/default/files/Documents/relative-contribution-vectors-introduction-translocation-invasive-marine-species.pdf

Hewitt, C. L., Campbell, M. L., Thresher, R. E., Martin, R. B., Boyd, S., Cohen, B. F., Currie, D. R., Gomon, M. F., Keough, M. J., Lewis, J. A., Lockett, M. M., Mays, N., McArthur, M. A., O'Hara, T. D., Poore, G. C. B., Ross, D. J., Storey, M. J., Watson, J. E., & Wilson, R. S. (2004). Introduced and cryptogenic species in Port Phillip Bay, Victoria, Australia. *Marine Biology, 144*(1), 183–202. https://doi.org/10.1007/s00227-003-1173-x

Hierro, J. L., Lortie, C. J., Villarreal, D., Estanga-Mollica, M. E., & Callaway, R. M. (2011). Resistance to *Centaurea solstitialis* invasion from annual and perennial grasses in California and Argentina. *Biological Invasions, 13*(10), 2249–2259. https://doi.org/10.1007/s10530-011-0037-4

Hierro, J. L., Villarreal, D., Eren, Ö., Graham, J. M., & Callaway, R. M. (2006). Disturbance facilitates invasion: The effects are stronger abroad than at home. *The American Naturalist, 168*(2), 144–156. https://doi.org/10.1086/505767

Hill, J. E. (2008). *Non-native species in aquaculture: Terminology, potential impacts, and the invasion process* (4303; p. 8). Southern Regional Aquaculture Center. http://www.takeaim.org/wp-content/uploads/2016/11/SRAC_4303.pdf

Hill, M. P., Bertelsmeier, C., Clusella-Trullas, S., Garnas, J., Robertson, M. P., & Terblanche, J. S. (2016). Predicted decrease in global climate suitability masks regional complexity of invasive fruit fly species response to climate change. In *Biological Invasions* (4; Vol. 18, Issue 4, pp. 1105–1119). SPRINGER. https://doi.org/10.1007/s10530-016-1078-5

Hill, M. P., Coetzee, J. A., Martin, G. D., Smith, R., & Strange, E. F. (2020). Invasive alien aquatic plants in South African freshwater ecosystems. In B. van Wilgen, J. Measey, D. Richardson, J. Wilson, & T. Zengeya (Eds.), *Biological Invasions in South Africa* (pp. 97–114). Springer, Cham. https://doi.org/10.1007/978-3-030-32394-3_4

Hill, M. P., Gallardo, B., & Terblanche, J. S. (2017). A global assessment of climatic niche shifts and human influence in insect invasions. *Global Ecology and Biogeography, 26*(6), 679–689. https://doi.org/10.1111/geb.12578

Hodgins, K. A., Bock, D. G., & Rieseberg, L. H. (2018). Trait evolution in invasive species. *Annual Plant Reviews, 1*(2), 1–37. https://doi.org/10.1002/9781119312994.apr0643

Hodgins, K. A., Rieseberg, L., & Otto, S. P. (2009). Genetic control of invasive plants species using selfish genetic elements. *Evolutionary Applications, 2*(4), 555–569. https://doi.org/10.1111/j.1752-4571.2009.00102.x

Hoekstra, J. M., Boucher, T. M., Ricketts, T. H., & Roberts, C. (2004). Confronting a biome crisis: Global disparities of habitat loss and protection: Confronting a biome crisis. *Ecology Letters, 8*(1), 23–29. https://doi.org/10.1111/j.1461-0248.2004.00686.x

Hoffmann, B. D., & Courchamp, F. (2016). Biological invasions and natural colonisations: Are they that different? *NeoBiota, 29*, 1–14. https://doi.org/10.3897/neobiota.29.6959

Hofstede, R. G. M., Groenendijk, J. P., Coppus, R., Fehse, J. C., & Sevink, J. (2002). Impact of pine plantations on soils and vegetation in the Ecuadorian high Andes. *Mountain Research and Development, 22*(2), 159–167. https://doi.org/10.1659/0276-4741(2002)022[0159:IOPPOS]2.0.CO;2

Holenstein, K., Simonson, W. D., Smith, K. G., Blackburn, T. M., & Charpentier, A. (2021). Non-native species surrounding protected areas influence the community of non-native species within them. *Frontiers in Ecology and Evolution, 8*, 625137. https://doi.org/10.3389/fevo.2020.625137

Holman, L. E., Hollenbeck, C. M., Ashton, T. J., & Johnston, I. A. (2019). Demonstration of the use of environmental DNA for the non-Invasive genotyping of a bivalve mollusk, the European Flat Oyster *(Ostrea edulis). Frontiers in Genetics, 10*, 1159. https://doi.org/10.3389/fgene.2019.01159

Holmes, P. M., Richardson, D. M., Esler, K. J., Witkowski, E. T. F., & Fourie, S. (2005). A decision-making framework for restoring riparian zones degraded by invasive alien plants in South Africa. *South African Journal of Science, 101*(11–12), 553. http://hdl.handle.net/11427/24115

Holway, D. A., Lach, L., Suarez, A. V., Tsutsui, N. D., & Case, T. J. (2002). The causes and consequences of ant invasions. *Annual Review of Ecology and Systematics, 33*(1), 181–233. https://doi.org/10.1146/annurev.ecolsys.33.010802.150444

Hood, W. G., & Naiman, R. J. (2000). Vulnerability of riparian zones to invasion by exotic vascular plants. *Plant Ecology, 148*(1), 105–114. https://doi.org/10.1023/A:1009800327334

Hovick, S. M., & Whitney, K. D. (2014). Hybridisation is associated with increased fecundity and size in invasive taxa: Meta-analytic support for the hybridisation-invasion hypothesis. *Ecology Letters, 17*(11), 1464–1477. https://doi.org/10.1111/ele.12355

Hovmøller, M. S., Walter, S., Bayles, R. A., Hubbard, A., Flath, K., Sommerfeldt, N., Leconte, M., Czembor, P., Rodriguez-Algaba, J., Thach, T., Hansen, J. G., Lassen, P., Justesen, A. F., Ali, S., & de Vallavieille-Pope, C. (2016). Replacement of the European wheat yellow rust population by new races from the centre of diversity in the near-Himalayan region. *Plant Pathology, 65*(3), 402–411. https://doi.org/10.1111/ppa.12433

Howard, G. W., & Chege, F. W. (2007). Invasions by plants in the inland waters and wetlands of Africa. In F. Gherardi (Ed.), *Biological invaders in inland waters: Profiles, distribution, and threats.* (Vol. 2, pp. 193–208). Springer. https://doi.org/10.1007/978-1-4020-6029-8_10

Howard, G. W., & Matindi, S. W. (2003). *Alien invasive species in Africa's wetlands—Some threats and solutions* (p. 15). IUCN Eastern Africa Regional Programme. https://portals.iucn.org/library/sites/library/files/documents/2003-003.pdf

Howarth, F. G. (1985). Impacts of alien land arthropods and mollusks on native plants and animals in Hawaii. *Hawaii's Terrestrial Ecosystems: Preservation and Management. University of Hawaii Press, Honolulu*, 149–179. http://www.hear.org/books/hte1985/pdfs/hte1985howarth.pdf

Howell, C., & Sawyer, J. W. D. (2006). *New Zealand naturalised vascular plant checklist*. New Zealand Plant conservation Network.

Hrivnák, R., Medvecká, J., Baláži, P., Bubíková, K., Oťaheľová, H., & Svitok, M. (2019). Alien aquatic plants in Slovakia over 130 years: Historical overview, current distribution and future perspectives. *NeoBiota, 49*, 37–56. https://doi.org/10.3897/neobiota.49.34318

Huang, D., Haack, R. A., & Zhang, R. (2011). Does global warming increase establishment rates of invasive alien species? A centurial time series analysis. *PLoS ONE, 6*(9), e24733. https://doi.org/10.1371/journal.pone.0024733

Hudson, J., Bourne, S. D., Seebens, H., Chapman, M. A., & Rius, M. (2022). The reconstruction of invasion histories with genomic data in light of differing levels of anthropogenic transport. *Philosophical Transactions of the Royal Society B: Biological Sciences, 377*(1846), 20210023. https://doi.org/10.1098/rstb.2021.0023

Hudson, J., Castilla, J. C., Teske, P. R., Beheregaray, L. B., Haigh, I. D., McQuaid, C. D., & Rius, M. (2021). Genomics-informed models reveal extensive stretches of coastline under threat by an ecologically dominant invasive species. *Proceedings of the National Academy of Sciences, 118*(23), e2022169118. https://doi.org/10.1073/pnas.2022169118

Hughes, A. C., Orr, M. C., Ma, K., Costello, M. J., Waller, J., Provoost, P., Yang, Q., Zhu, C., & Qiao, H. (2021). Sampling biases shape our view of the natural world. *Ecography, 44*(9), 1259–1269. https://doi.org/10.1111/ecog.05926

Hughes, K. A., & Convey, P. (2012). Determining the native/non-native status of newly discovered terrestrial and freshwater species in Antarctica—Current knowledge, methodology and management action. *Journal of Environmental Management, 93*(1), 52–66. https://doi.org/10.1016/j.jenvman.2011.08.017

Hughes, K. A., Convey, P., Maslen, N. R., & Smith, R. I. L. (2010). Accidental transfer of non-native soil organisms into Antarctica on construction vehicles. *Biological Invasions, 12*(4), 875–891. https://doi.org/10.1007/s10530-009-9508-2

Hughes, K. A., Cowan, D. A., & Wilmotte, A. (2015). Protection of Antarctic microbial communities – 'out of sight, out of mind'. *Frontiers in Microbiology, 6*, 151. https://doi.org/10.3389/fmicb.2015.00151

Hughes, K. A., Lee, J. E., Tsujimoto, M., Imura, S., Bergstrom, D. M., Ware, C., Lebouvier, M., Huiskes, A. H. L., Gremmen, N. J. M., Frenot, Y., Bridge, P. D., & Chown, S. L. (2011). Food for thought: Risks of non-native species transfer to the Antarctic region with fresh produce. *Biological Conservation, 144*(5), 1682–1689. https://doi.org/10.1016/j.biocon.2011.03.001

Hughes, K. A., & Pertierra, L. R. (2016). Evaluation of non-native species policy development and implementation within the Antarctic Treaty area. *Biological Conservation, 200*, 149–159. https://doi.org/10.1016/j.biocon.2016.03.011

Hughes, K. A., Pertierra, L. R., Molina-Montenegro, M. A., & Convey, P. (2015). Biological invasions in terrestrial Antarctica: What is the current status and can we respond? *Biodiversity and Conservation, 24*(5), 1031–1055. https://doi.org/10.1007/s10531-015-0896-6

Hughes, K. A., Pescott, O. L., Peyton, J. M., Adriaens, T., Cottier-Cook, E. J., Key, G., Rabitsch, W., Tricarico, E., Barnes, D. K. A., Baxter, N., Belchier, M., Blake, D., Convey, P., Dawson, W., Frohlich, D., Gardiner, L. M., González-Moreno, P., James, R., Malumphy, C., … Roy, H. E. (2020). Invasive non-native species likely to threaten biodiversity and ecosystems in the Antarctic Peninsula region. *Global Change Biology, 26*(4), 2702–2716. https://doi.org/10.1111/gcb.14938

Huiskes, A. H. L., Gremmen, N. J. M., Bergstrom, D. M., Frenot, Y., Hughes, K. A., Imura, S., Kiefer, K., Lebouvier, M., Lee, J. E., Tsujimoto, M., Ware, C., Van de Vijver, B., & Chown, S. L. (2014). Aliens in Antarctica: Assessing transfer of plant propagules by human visitors to reduce invasion risk. *Biological Conservation, 171*, 278–284. https://doi.org/10.1016/j.biocon.2014.01.038

Hulme, P. E. (2009). Trade, transport and trouble: Managing invasive species pathways in an era of globalization. *Journal of Applied Ecology, 46*(1), 10–18. https://doi.org/10.1111/j.1365-2664.2008.01600.x

Hulme, P. E. (2011). Addressing the threat to biodiversity from botanic gardens. *Trends in Ecology and Evolution, 26*(4), 168–174. https://doi.org/10.1016/j.tree.2011.01.005

Hulme, P. E. (2014). Invasive species challenge the global response to emerging diseases. *Trends in Parasitology, 30*(6), 267–270. https://doi.org/10.1016/j.pt.2014.03.005

Hulme, P. E. (2017). Climate change and biological invasions: Evidence, expectations, and response options. *Biological Reviews, 92*(3), 1297–1313. https://doi.org/10.1111/brv.12282

Hulme, P. E. (2021). Unwelcome exchange: International trade as a direct and indirect driver of biological invasions worldwide. *One Earth, 4*(5), 666–679. https://doi.org/10.1016/j.oneear.2021.04.015

Hulme, P. E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., Nentwig, W., Olenin, S., Panov, V., Pergl, J., Pyšek, P., Roques, A., Sol, D., Solarz, W., & Vilà, M. (2008). Grasping at the routes of biological invasions: A framework for integrating pathways into policy. *Journal of Applied Ecology, 45*(2), 403–414. https://doi.org/10.1111/j.1365-2664.2007.01442.x

Humber, J. M., & Hermanutz, L. (2011). Impacts of non-native plant and animal invaders on gap regeneration in a protected boreal forest. *Biological Invasions, 13*(10), 2361–2377. https://doi.org/10.1007/s10530-011-0048-1

Hunter, M. E., Oyler-McCance, S. J., Dorazio, R. M., Fike, J. A., Smith, B. J., Hunter, C. T., Reed, R. N., & Hart, K. M. (2015). Environmental DNA (eDNA) sampling improves occurrence and detection estimates of invasive Burmese pythons. *PLoS ONE, 10*(4), e0121655. https://doi.org/10.1371/journal.pone.0121655

Hurley, B. P., Garnas, J., Wingfield, M. J., Branco, M., Richardson, D. M., & Slippers, B. (2016). Increasing numbers and intercontinental spread of invasive insects on eucalypts. *Biological Invasions, 18*(4), 921–933. https://doi.org/10.1007/s10530-016-1081-x

Hussner, A. (2012). Alien aquatic plant species in European countries. *Weed Research, 52*(4), 297–306. https://doi.org/10.1111/j.1365-3180.2012.00926.x

Hussner, A., Van de Weyer, K., Gross, E. M., & Hilt, S. (2010). Comments on increasing number and abundance of non-indigenous aquatic macrophyte species in Germany. *Weed Research, 50*(6), 519–526. https://doi.org/10.1111/j.1365-3180.2010.00812.x

Hutchinson, J., Reise, H., & Robinson, D. (2014). A biography of an invasive terrestrial slug: The spread, distribution and habitat of *Deroceras invadens*. *NeoBiota, 23*, 17–64. https://doi.org/10.3897/neobiota.23.7745

ICES. (2005). *The intentional introduction of the marine red king crab Paralithodes camtschaticus into the Southern Barents Sea* (ICES Cooperative Research Report No. 277; p. 18). International Council for the Exploration of the Sea.

ICES. (2019). *ICES VIEWPOINT: Biofouling on vessels – what is the risk, and what might be done about it?* [Report of the ICES Advisory Committee]. International Council for the Exploration of the Sea. https://doi.org/10.17895/ices.advice.4687

Ihlow, F., Vamberger, M., Flecks, M., Hartmann, T., Cota, M., Makchai, S., Meewattana, P., Dawson, J. E., Kheng, L., Rödder, D., & Fritz, U. (2016). Integrative Taxonomy of Southeast Asian Snail-Eating Turtles (Geoemydidae: *Malayemys*) Reveals a New Species and Mitochondrial Introgression. *PLoS ONE, 11*(4), e0153108. https://doi.org/10.1371/journal.pone.0153108

Imada, C. T. (2012). *Hawaiian native and naturalized vascular plants checklist* (Bishop Musem Technical Report 60; p. 380). Bishop Museum. http://www.aecos.com/AECOS/EG%20Files/eg2012002.pdf

Inderjit, Pergl, J., van Kleunen, M., Hejda, M., Babu, C. R., Majumdar, S., Singh, P., Singh, S. P., Salamma, S., Rao, B. R. P., & Pyšek, P. (2018). Naturalized alien flora of the Indian states: Biogeographic patterns, taxonomic structure and drivers of species richness. *Biological Invasions, 20*(6), 1625–1638. https://doi.org/10.1007/s10530-017-1622-y

Innocenti, G., Stasolla, G., Goren, M., Stern, N., Levitt-Barmats, Y., Diamant, A., & Galil, B. S. (2017). Going down together: Invasive host, *Charybdis longicollis* (Decapoda: Brachyura: Portunidae) and invasive parasite, *Heterosaccus dollfusi* (Cirripedia: Rhizocephala: Sacculinidae) on the upper slope off the Mediterranean coast of Israel. *Marine Biology Research, 13*(2), 229–236. https://doi.org/10.1080/17451000.2016.1240873

IPBES. (2016). *The assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3402857

IPBES. (2018a). *The IPBES regional assessment report on biodiversity and ecosystem services for Africa.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://zenodo.org/record/3236178#.YmjyldrP2Uk

IPBES. (2018b). *The IPBES regional assessment report on biodiversity and ecosystem services for Asia and the Pacific* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237374

IPBES. (2018c). *The IPBES regional assessment report on biodiversity and ecosystem services for Europe and Central Asia* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237429

IPBES. (2019a). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3831673

IPBES. (2019b). *IPBES Units of Analysis* (Version 1) [dataset]. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3975694

IPBES Technical Support Unit On Knowledge And Data. (2021). *IPBES regions and sub-regions* (1.2) [dataset]. Zenodo. https://doi.org/10.5281/ZENODO.5719431

IPCC. (2019). *Summary for Policymakers* (IPCC Special Report on the Ocean and Cryosphere in a Changing Climate). Intergovernmental Panel on Climate Change. https://doi.org/10.1017/9781009157964.001

Isbell, F., Craven, D., Connolly, J., Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T. M., Bonin, C., Bruelheide, H., de Luca, E., Ebeling, A., Griffin, J. N., Guo, Q., Hautier, Y., Hector, A., Jentsch, A., Kreyling, J., Lanta, V., Manning, P., … Eisenhauer, N. (2015). Biodiversity increases the resistance of ecosystem productivity to climate extremes. *Nature, 526*(7574), 574–577. https://doi.org/10.1038/nature15374

Islam, Md. M., Amin, A. S. M. R., & Sarker, S. K. (2003). Bangladesh. In N. Pallewatta, J. K. Reaser, & A. T. Gutierrez (Eds.), *Invasive Alien Species in South-Southeast Asia: National Reports & Directory of Resources.* Global Invasive Species Programme.

Iwasaki, K. (2006). Human-mediated introduction of marine organisms in Japan: A review. Assessment and control of biological invasion risks. In F. Koike, M. N. Clout, M. Kawamichi, M. De Poorter, & K. Iwatsuki (Eds.), *Assessment and Control of Biological Invasion Risks* (pp. 104–112). SHOUKADOH Book Sellers.

Jack, C. N., Petipas, R. H., Cheeke, T. E., Rowland, J. L., & Friesen, M. L. (2021). Microbial Inoculants: Silver Bullet or Microbial Jurassic Park? *Trends in Microbiology, 29*(4), 299–308. https://doi.org/10.1016/j.tim.2020.11.006

Jackson, D., Drumm, A., McEvoy, S., Jensen, Ø., Mendiola, D., Gabiña, G., Borg, J. A., Papageorgiou, N., Karakassis, Y., & Black, K. D. (2015). A pan-European valuation of the extent, causes and cost of escape events from sea cage fish farming. *Aquaculture, 436*, 21–26. https://doi.org/10.1016/j.aquaculture.2014.10.040

Jackson, M. C., & Grey, J. (2013). Accelerating rates of freshwater invasions in the catchment of the River Thames. *Biological Invasions, 15*(5), 945–951. https://doi.org/10.1007/s10530-012-0343-5

Janick, J. (2007). Plant Exploration: From Queen Hatshepsut to Sir Joseph Banks. *HortScience, 42*(2), 191–196. https://doi.org/10.21273/HORTSCI.42.2.191

Janion-Scheepers, C., & Griffiths, C. L. (2020). Alien Terrestrial Invertebrates in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 185–205). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_7

Jarić, I., Courchamp, F., Correia, R. A., Crowley, S. L., Essl, F., Fischer, A., González-Moreno, P., Kalinkat, G., Lambin, X., Lenzner, B., Meinard, Y., Mill, A., Musseau, C., Novoa, A., Pergl, J., Pyšek, P., Pyšková, K., Robertson, P., Schmalensee, M., … Jeschke, J. M. (2020). The role of species charisma in biological invasions. *Frontiers in Ecology and the Environment, 18*(6), 345–353. https://doi.org/10.1002/fee.2195

Jarić, I., Heger, T., Castro Monzon, F., Jeschke, J. M., Kowarik, I., McConkey, K. R., Pyšek, P., Sagouis, A., & Essl, F. (2019). Crypticity in Biological Invasions. *Trends in Ecology & Evolution, 34*(4), 291–302. https://doi.org/10.1016/j.tree.2018.12.008

Jaryan, V., Uniyal, S. K., Gupta, R. C., & Singh, R. D. (2013). Alien flora of Indian Himalayan state of Himachal Pradesh. *Environmental Monitoring and Assessment, 185*(7), 6129–6153. https://doi.org/10.1007/s10661-012-3013-2

Jeney, Zs., & Jeney, G. (1995). Recent achievements in studies on diseases of common carp (*Cyprinus carpio* L.). *Aquaculture, 129*(1), 397–420. https://doi.org/10.1016/0044-8486(94)00283-T

Jeschke, J. M., Aparicio, L. G., Haider, S., Heger, T., Lortie, C. J., Pyšek, P., & Strayer, D. L. (2012). Taxonomic bias and lack of cross-taxonomic studies in invasion biology. *Frontiers in Ecology and the Environment, 10*(7), 349–350. https://doi.org/10.1890/12.WB.016

Johnston, M. A., Nuttall, M. F., Eckert, R. J., Embesi, J. A., Slowey, N. C., Hickerson, E. L., & Schmahl, G. P. (2013). *Long-Term Monitoring at the East and West Flower Garden Banks National Marine Sanctuary, 2009–2010* (Technical Report Volume 1; p. 218). U.S. Department of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region.

Jones, C. G., Lawton, J. H., & Shachak, M. (1994). Organisms as Ecosystem Engineers. In F. B. Samson & F. L. Knopf (Eds.), *Ecosystem Management* (pp. 130–147). Springer. https://doi.org/10.1007/978-1-4612-4018-1_14

Jones, C. W., Risi, M. M., Cleeland, J., & Ryan, P. G. (2019). First evidence of mouse attacks on adult albatrosses and petrels breeding on sub-Antarctic Marion and Gough Islands. *Polar Biology, 42*, 619–623. https://doi.org/10.1007/s00300-018-02444-6

Jones, D. R. (2003). Plant viruses transmitted by whiteflies. *European Journal of Plant Pathology, 109*(3), 195–219. https://doi.org/10.1023/A:1022846630513

Jones, E. J., Kraaij, T., Fritz, H., & Moodley, D. (2019). A global assessment of terrestrial alien ferns (Polypodiophyta): Species' traits as drivers of naturalisation and invasion. *Biological Invasions, 21*(3), 861–873. https://doi.org/10.1007/s10530-018-1866-1

Jones, E., & Thornber, C. S. (2010). Effects of habitat-modifying invasive macroalgae on epiphytic algal communities. *Marine Ecology Progress Series, 400*, 87–100. https://doi.org/10.3354/meps08391

Jones, H. P., Holmes, N. D., Butchart, S. H. M., Tershy, B. R., Kappes, P. J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D. P., Bonnaud, E., Burbidge, A. A., Campbell, K., Courchamp, F., Cowan, P. E., Cuthbert, R. J., Ebbert, S., Genovesi, P., Howald, G. R., Keitt, B. S., Kress, S. W., … Croll, D. A. (2016). Invasive mammal eradication on islands results in substantial conservation gains. *Proceedings of the National Academy of Sciences, 113*(15), 4033–4038. https://doi.org/10.1073/pnas.1521179113

Jørgensen, L. L., & Nilssen, E. M. (2011). The Invasive History, Impact and Management of the Red King Crab *Paralithodes camtschaticus* off the Coast of Norway. In B. S. Galil, P. F. Clark, & J. T. Carlton (Eds.), *In the Wrong Place—Alien Marine Crustaceans: Distribution, Biology and Impacts* (Vol. 6, pp. 521–536). Springer Netherlands. https://doi.org/10.1007/978-94-007-0591-3_18

Ju, R.-T., Li, X., Jiang, J.-J., Wu, J., Liu, J., Strong, D. R., & Li, B. (2020). Emerging risks of non-native species escapes from aquaculture: Call for policy improvements in China and other developing countries. *Journal of Applied Ecology, 57*(1), 85–90. https://doi.org/10.1111/1365-2664.13521

Jukonienė, I., Dobravolskaitė, R., Sendžikaitė, J., Skipskytė, D., & Repečkienė, J. (2015). Disturbed peatlands as a habitat of an invasive moss *Campylopus introflexus* in Lithuania. *Boreal Environment Research, 20*, 724–734. http://hdl.handle.net/10138/228307

Jung, T., Horta Jung, M., Webber, J. F., Kageyama, K., Hieno, A., Masuya, H., Uematsu, S., Pérez-Sierra, A., Harris, A. R., Forster, J., Rees, H., Scanu, B., Patra, S., Kudláček, T., Janoušek, J., Corcobado, T., Milenković, I., Nagy, Z., Csorba, I., … Brasier, C. M. (2021). The Destructive Tree Pathogen *Phytophthora ramorum* Originates from the Laurosilva Forests of East Asia. *Journal of Fungi, 7*(3), 226. https://doi.org/10.3390/jof7030226

Justine, J.-L., Winsor, L., Gey, D., Gros, P., & Thévenot, J. (2020). *Obama chez moi!* The invasion of metropolitan France by the land planarian *Obama nungara* (Platyhelminthes, Geoplanidae). *PeerJ, 8*, e8385. https://doi.org/10.7717/peerj.8385

Kairo, M., Ali, B., Cheesman, O., Haysom, K., & Murphy, S. (2003). *Invasive species threats in the Caribbean Region* (Report to the Nature Conservancy, Arlington). Centre for Agriculture and Bioscience International. https://caribbeaninvasives.org/wp-content/uploads/2020/12/caribbean_invasives_paper.pdf

Kalusová, V., Chytrý, M., van Kleunen, M., Mucina, L., Dawson, W., Essl, F., Kreft, H., Pergl, J., Weigelt, P., Winter, M., & Pyšek, P. (2017). Naturalization of European plants on other continents: The role of donor habitats. *Proceedings of the National Academy of Sciences, 114*(52), 13756–13761. https://doi.org/10.1073/pnas.1705487114

Kalwij, J. M., Robertson, M. P., & van Rensburg, B. J. (2008). Human activity facilitates altitudinal expansion of exotic plants along a road in montane grassland, South Africa. *Applied Vegetation Science, 11*(4), 491–498. https://doi.org/10.3170/2008-7-18555

Kalwij, J. M., Robertson, M. P., & van Rensburg, B. J. (2015). Annual monitoring reveals rapid upward movement of exotic plants in a montane ecosystem. *Biological Invasions, 17*(12), 3517–3529. https://doi.org/10.1007/s10530-015-0975-3

Kamp, J., Koshkin, M. A., Bragina, T. M., Katzner, T. E., Milner-Gulland, E. J., Schreiber, D., Sheldon, R., Shmalenko, A., Smelansky, I., Terraube, J., & Urazaliev, R. (2016). Persistent and novel threats to the biodiversity of Kazakhstan's steppes and semi-deserts. *Biodiversity and Conservation, 25*(12), 2521–2541. USGS Publications Warehouse. https://doi.org/10.1007/s10531-016-1083-0

Kannan, R., Shackleton, C. M., Krishnan, S., & Shaanker, R. U. (2016). Can local use assist in controlling invasive alien species in tropical forests? The case of *Lantana camara* in southern India. *Forest Ecology and Management, 376*, 166–173. https://doi.org/10.1016/j.foreco.2016.06.016

Kappel, A. P., Trotter, R. T., Keena, M. A., Rogan, J., & Williams, C. A. (2017). Mapping of the Asian longhorned beetle's time to maturity and risk to invasion at contiguous United States extent. *Biological Invasions, 19*(7), 1999–2013. https://doi.org/10.1007/s10530-017-1398-0

Kark, S., Solarz, W., Chiron, F., Clergeau, P., & Shirley, S. M. (2009). Alien Birds, Amphibians and Reptiles of Europe. In DAISIE (Ed.), *Handbook of Alien Species in Europe* (pp. 105–118). Springer. https://doi.org/10.1007/978-1-4020-8280-1_8

Kartesz, J. T. (2014). *The Biota of North America Program (BONAP): North American Vascular Flora.* Taxonomic Data Center. (http://www.bonap.net/tdc). Chapel Hill, N.C. http://www.bonap.net/tdc

Kassie, M., Wossen, T., De Groote, H., Tefera, T., Sevgan, S., & Balew, S. (2020). Economic impacts of fall armyworm and its management strategies: Evidence from southern Ethiopia. *European Review of Agricultural Economics, 47*(4), 1473–1501. https://doi.org/10.1093/erae/jbz048

Katsanevakis, S., Zenetos, A., Belchior, C., & Cardoso, A. C. (2013). Invading European Seas: Assessing pathways of introduction of marine aliens. *Ocean & Coastal Management, 76*, 64–74. https://doi.org/10.1016/j.ocecoaman.2013.02.024

Katsanevakis, S., Zenetos, A., Corsini-Foka, M., & Tsiamis, K. (2020). Biological Invasions in the Aegean Sea: Temporal Trends, Pathways, and Impacts. In *The Handbook of Environmental Chemistry.* Springer Berlin Heidelberg. https://doi.org/10.1007/698_2020_642

Keeley, J. E., Baer-Keeley, M., & Fotheringham, C. J. (2005). Alien Plant Dynamics Following Fire in Mediterranean-Climate California Shrublands. *Ecological Applications, 15*(6), 2109–2125. https://doi.org/10.1890/04-1222

Keith, P. (2002). Freshwater fish and decapod crustacean populations on Réunion island, with an assessment of species introductions. *Bulletin Français de La Pêche et de La Pisciculture, 364*, 97–107. https://doi.org/10.1051/kmae:2002005

Keller, R. P., Ermgassen, P. S. E. Z., & Aldridge, D. C. (2009). Vectors and Timing of Freshwater Invasions in Great Britain. *Conservation Biology, 23*(6), 1526–1534. https://doi.org/10.1111/j.1523-1739.2009.01249.x

Keller, S. R., & Taylor, D. R. (2008). History, chance and adaptation during biological invasion: Separating stochastic phenotypic evolution from response to selection. *Ecology Letters, 11*(8), 852–866. https://doi.org/10.1111/j.1461-0248.2008.01188.x

Keller, S. R., & Taylor, D. R. (2010). Genomic admixture increases fitness during a biological invasion. *Journal of Evolutionary Biology, 23*(8), 1720–1731. https://doi.org/10.1111/j.1420-9101.2010.02037.x

Kenis, M., Auger-Rozenberg, M.-A., Roques, A., Timms, L., Péré, C., Cock, M. J. W., Settele, J., Augustin, S., & Lopez-Vaamonde, C. (2009). Ecological effects of invasive alien insects. *Biological Invasions, 11*(1), 21–45. https://doi.org/10.1007/s10530-008-9318-y

Kenis, M., Rabitsch, W., Auger-Rozenberg, M.-A., & Roques, A. (2007). How can alien species inventories and interception data help us prevent insect invasions? *Bulletin of Entomological Research, 97*(5), 489–502. https://doi.org/10.1017/S0007485307005184

Keppel, G., Morrison, C., Meyer, J.-Y., & Boehmer, H. J. (2014). Isolated and vulnerable: The history and future of Pacific Island terrestrial biodiversity. *Pacific Conservation Biology, 20*(2), 136–145. https://doi.org/10.1071/PC140136

Kerckhof, F., De Mesel, I., & Degraer, S. (2018). First European record of the invasive barnacle *Balanus glandula* Darwin, 1854. *BioInvasions Records, 7*(1), 21–31. https://doi.org/10.3391/bir.2018.7.1.04

Kestrup, Å. M., Smith, D. L., & Therriault, T. W. (Eds.). (2015). *Report of Working Group 21 on Non-indigenous Aquatic Species. PICES Scientific Report No. 48.* North Pacific Marine Science Organization (PICES). www.pices.int/publications/scientific_reports/default.aspx

Khlyap, L. A., Warshavsky, A. A., & Bobrov, V. V. (2011). Diversity of alien mammal species in different regions of Russia. *Russian Journal of Biological Invasions, 2*(4), 293–299. https://doi.org/10.1134/S2075111711040059

Khuroo, A. A., Reshi, Z. A., Malik, A. H., Weber, E., Rashid, I., & Dar, G. H. (2012). Alien flora of India: Taxonomic composition, invasion status and biogeographic affiliations. *Biological Invasions, 14*(1), 99–113. https://doi.org/10.1007/s10530-011-9981-2

Khuroo, A. A., Weber, E., Malik, A. H., Reshi, Z. A., & Dar, G. H. (2011). Altitudinal distribution patterns of the native and alien woody flora in Kashmir Himalaya, India. *Environmental Research, 111*(7), 967–977. https://doi.org/10.1016/j.envres.2011.05.006

Kier, G., Kreft, H., Lee, T. M., Jetz, W., Ibisch, P. L., Nowicki, C., Mutke, J., & Barthlott, W. (2009). A global assessment of endemism and species richness across island and mainland regions. *Proceedings of the National Academy of Sciences, 106*(23), 9322–9327. https://doi.org/10.1073/pnas.0810306106

Kipp, R., Bailey, S. A., MacIsaac, H. J., & Ricciardi, A. (2010). Transoceanic ships as vectors for nonindigenous freshwater bryozoans. *Diversity and Distributions, 16*(1), 77–83. https://doi.org/10.1111/j.1472-4642.2009.00629.x

Kiritani, K., & Morimoto, N. (2004). Invasive Insect and Nematode Pests from North America. *Global Environmental Research, 8*(1), 75–88. https://www.researchgate.net/publication/241529403_Invasive_Insect_and_Nematode_Pests_from_North_America

Kiritani, K., & Yamamura, K. (2003). Exotic insects and their pathways for invasion. In G. M. Ruiz & J. T. Carlton (Eds.), *Invasive species-vectors and management strategies* (pp. 44–67). Island Press.

Kirk, P. (2021). *Species Fungorum Plus: Fungi.* Catalogue of Life Checklist. https://www.catalogueoflife.org/data/taxon/F

Kirkpatric, W., Page, A., & Massam, M. (2008). *European Rabbit* (Oryctolagus cuniculus) *risk assessment for Australia.* Department of Agriculture and Food, West Australia.

Kistner-Thomas, E. J. (2019). The Potential Global Distribution and Voltinism of the Japanese Beetle (Coleoptera: Scarabaeidae) Under Current and Future Climates. *Journal of Insect Science, 19*(2). https://doi.org/10.1093/jisesa/iez023

Kitano, S. (2004). Ecological Impacts of Rainbow, Brown and Brook Trout in Japanese Inland Waters. *Global Environmental Research, 8*(1), 41–50. https://sakhtaimen.ru/userfiles/Library/English/kitano_2004.pdf

KLHRI. (2003). *Penyebaran jenis tumbuhan asing di Indonesia.* Southeast Asian Regional Centre for Tropical Biology.

Klinger, T., Padilla, D. K., & Britton-Simmons, K. (2006). Two invaders achieve higher densities in reserves. *Aquatic Conservation-Marine and Freshwater Ecosystems, 16*(3), 301–311. https://doi.org/10.1002/aqc.717

Klotz, S., Kühn, I., & Durka, W. (2003). *BIOFLOR – Eine Datenbank mit biologisch-ökologischen Merkmalen zur Flora von Deutschland* (Vol. 38). Landwirtschaftsvlg Münster.

Knapp, C. R., Iverson, J. B., Buckner, S. D., & Cant, S. V. (2011). Conservation of amphibians and reptiles in The Bahamas. In A. Hailey, B. S. Wilson, & J. A. Horrocks (Eds.), *Conservation of Caribbean Island Herpetofaunas, Vol 2: Regional Accounts of the West Indies* (pp. 53–87). Brill. https://doi.org/10.1163/ej.9789004194083.i-439.21

Knapp, S., Kühn, I., Stolle, J., & Klotz, S. (2010). Changes in the functional composition of a Central European urban flora over three centuries. *Perspectives in Plant Ecology, Evolution and Systematics, 12*(3), 235–244. https://doi.org/10.1016/j.ppees.2009.11.001

Kostecki, C., Rochette, S., Girardin, R., Blanchard, M., Desroy, N., & Le Pape, O. (2011). Reduction of flatfish habitat as a consequence of the proliferation of an invasive mollusc. *Estuarine, Coastal and Shelf Science, 92*(1), 154–160. https://doi.org/10.1016/j.ecss.2010.12.026

Kouba, A., Petrusek, A., & Kozák, P. (2014). Continental-wide distribution of crayfish species in Europe: Update and maps. *Knowledge and Management of Aquatic Ecosystems, 413*, 05. https://doi.org/10.1051/kmae/2014007

Kowarik, I. (1990). Some responses of flora and vegetation to urbanization in central Europe. In H. Sukopp, S. Hejnỳ, & I. Kowarik (Eds.), *Urban Ecology: Plants and Plant Communities in Urban Environments* (pp. 45–74). SPB Academic Publishing.

Kowarik, I. (1995a). On the role of alien species in urban flora and vegetation. In P. Pyšek, K. Prach, M. Rejmánek, & P. Wade (Eds.), *Plant invasions: General aspects and special problems* (pp. 85–103). SPB Academic Publishing.

Kowarik, I. (1995b). Time lags in biological invasions with regard to the success and failure of alien species. In P. Pyšek, K. Prach, M. Rejmánek, & M. Wade (Eds.), *Plant invasions: General aspects and special problems* (pp. 15–38). SPB Academic Publishing.

Kozhevnikov, A. E., & Kozhevnikova, Z. V. (2011). Alien species plant complex as a component of the Far East of Russia natural flora: Diversity and regional changes of taxonomical structure. In *V.L. Komarov's Memorial Lectures* (Vol. 58, pp. 5–31).

Kramer, A. M., Annis, G., Wittmann, M. E., Chadderton, W. L., Rutherford, E. S., Lodge, D. M., Mason, L., Beletsky, D., Riseng, C., & Drake, J. M. (2017). Suitability of Laurentian Great Lakes for invasive species based on global species distribution models and local habitat. *Ecosphere, 8*(7), e01883. https://doi.org/10.1002/ecs2.1883

Kraus, F. (2009). *Alien reptiles and amphibians: A scientific compendium and analysis* (Vol. 4). Springer. https://doi.org/10.1007/978-1-4020-8946-6

Kraus, F. (2011). Reptiles and amphibians. In D. Simberloff & M. Rejmanek (Eds.), *Encyclopedia of Biological Invasions* (pp. 590–594). University of California Press, Berkeley and Los Angeles.

Kriticos, D. J., & Brunel, S. (2016). Assessing and managing the current and future pest risk from water hyacinth, *(Eichhornia crassipes)*, an invasive aquatic plant threatening the environment and water security. *PLoS ONE, 11*(8), e0120054. https://doi.org/10.1371/journal.pone.0120054

Kriticos, D. J., Kean, J. M., Phillips, C. B., Senay, S. D., Acosta, H., & Haye, T. (2017). The potential global distribution of the brown marmorated stink bug, *Halyomorpha halys*, a critical threat to plant biosecurity. *Journal of Pest Science, 90*(4), 1033–1043. https://doi.org/10.1007/s10340-017-0869-5

Kriticos, D. J., Morin, L., Leriche, A., Anderson, R. C., & Caley, P. (2013). Combining a Climatic Niche Model of an Invasive Fungus with Its Host Species Distributions to Identify Risks to Natural Assets: *Puccinia psidii* Sensu Lato in Australia. *PLoS ONE, 8*(5), e64479. https://doi.org/10.1371/journal.pone.0064479

Kroschel, J., Sporleder, M., Tonnang, H. E. Z., Juarez, H., Carhuapoma, P., Gonzales, J. C., & Simon, R. (2013). Predicting climate-change-caused Changes in global temperature on potato tuber moth *Phthorimaea operculella* (Zeller) distribution and abundance using phenology modeling and GIS mapping. *Agricultural and Forest Meteorology, 170*, 228–241. https://doi.org/10.1016/j.agrformet.2012.06.017

Krueger-Hadfield, S. A., Kollars, N. M., Strand, A. E., Byers, J. E., Shainker, S. J., Terada, R., Greig, T. W., Hammann, M., Murray, D. C., Weinberger, F., & Sotka, E. E. (2017). Genetic identification of source and likely vector of a widespread marine invader. *Ecology and Evolution, 7*(12), 4432–4447. https://doi.org/10.1002/ece3.3001

Krysko, K. L., Burgess, J. P., Rochford, M. R., Gillette, C. R., Cueva, D., Enge, K. M., Somma, L. A., Stabile, J. L., Smith, D. C., Wasilewski, J. A., Kieckhefer III, G. N., Granatosky, M. C., & Nielsen, S. V. (2011). Verified non-indigenous amphibians and reptiles in Florida from 1863 through 2010: Outlining the invasion process and identifying invasion pathways and stages. *Zootaxa, 3028*(1), 1. https://doi.org/10.11646/zootaxa.3028.1.1

Krysko, K. L., Somma, L. A., Smith, D. C., Gillette, C. R., Cueva, D., Wasilewski, J. A., Enge, K. M., Johnson, S. A., Campbell, T. S., Edwards, J. R., Rochford, M. R., Tompkins, R., Fobb, J. L., Mullin, S., Lechowicz, C., Hazelton, D., & Warren, A. (2016). New verified nonindigenous amphibians and reptiles in Florida, 1976 through 2015, with a summary of over 152 years of introductions. *Reptiles & Amphibians, 23*(2), 110–143. https://doi.org/10.17161/randa.v23i2.14119

Ksenofontov, S., Backhaus, N., & Schaepman-Strub, G. (2019). 'There are new species': Indigenous knowledge of biodiversity change in Arctic Yakutia. *Polar Geography, 42*(1), 34–57. https://doi.org/10.1080/1088937X.2018.1547326

Kuebbing, S. E., McCary, M. A., Lieurance, D., Nuñez, M. A., Chiuffo, M. C., Zhang, B., Seebens, H., Simberloff, D., & Meyerson, L. A. (2022). A self-study of editorial board diversity at Biological Invasions. *Biological Invasions, 24*(2), 321–332. https://doi.org/10.1007/s10530-021-02664-8

Kueffer, C., McDougall, K., Alexander, J., Daehler, C., Edwards, P., Haider, S., Milbau, A., Parks, C., Pauchard, A., Reshi, Z. A., Rew, L. J., Schroder, M., & Seipel, T. (2013). Plant Invasions into Mountain Protected Areas: Assessment, Prevention and Control at Multiple Spatial Scales. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7, pp. 89–113). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_6

Kueffer, C., Vos, P., Lavergne, C., & Mauremootoo, J. (2004). Case studies on the status of invasive woody plant species in the Western Indian Ocean. 1. Synthesis. *Forest Health and Biosecurity Working Paper, No.4-1E*. https://www.cabdirect.org/cabdirect/abstract/20123338431

Kuhman, T. R., Pearson, S. M., & Turner, M. G. (2010). Effects of land-use history and the contemporary landscape on non-native plant invasion at local and regional scales in the forest-dominated southern Appalachians. *Landscape Ecology, 25*(9), 1433–1445. https://doi.org/10.1007/s10980-010-9500-3

Kumar, A. B. (2000). Exotic fishes and freshwater fish diversity. *Zoo's Print Journal, 15*(11), 363–367. https://www.zoosprint.zooreach.org/index.php/zpj/article/view/5881

Kumar, S., & Mathur, M. (2014). Impact of invasion by *Prosopis juliflora* on plant communities in arid grazing lands. *Tropical Ecology, 55*(1), 33–47.

Kurniawan, R., & Paramita, I. G. A. A. P. (2020). List of aquatic plants at several priority lakes for conservation in Indonesia. *IOP Conference Series: Earth and Environmental Science, 535*(1), 012055. https://doi.org/10.1088/1755-1315/535/1/012055

Kutz, S. J., Hoberg, E. P., Nagy, J., Polley, L., & Elkin, B. (2004). "Emerging" Parasitic Infections in Arctic Ungulates. *Integrative and Comparative Biology, 44*(2), 109–118. https://doi.org/10.1093/icb/44.2.109

La Sorte, F. A., Aronson, M. F. J., Williams, N. S. G., Celesti-Grapow, L., Cilliers, S., Clarkson, B. D., Dolan, R. W., Hipp, A., Klotz, S., Kühn, I., Pyšek, P., Siebert, S., & Winter, M. (2014). Beta diversity of urban floras among European and non-European cities: Beta diversity of urban floras. *Global Ecology and Biogeography, 23*(7), 769–779. https://doi.org/10.1111/geb.12159

Lambdon, P. W., Pyšek, P., Başnou, C., Hejda, M., Arianoutsou, M., Essl, F., Jarošík, V., Pergl, J., Winter, M., Anastasiu, P., Andriopoulos, P., Bazos, I., Brundu, G., Celesti-Grapow, L., Chassot, P., Delipetrou, P., Josefsson, M., Kark, S., Klotz, S., … Hulme, P. E. (2008). Alien flora of Europe: Species diversity, temporal trends, geographical patterns and research needs. *Preslia, 80*(2), 101–149. http://www.preslia.cz/P082Lam.pdf

Lang, A. C., & Buschbaum, C. (2010). Facilitative effects of introduced Pacific oysters on native macroalgae are limited by a secondary invader, the seaweed *Sargassum muticum. Journal of Sea Research, 63*(2), 119–128. https://doi.org/10.1016/j.seares.2009.11.002

Langdon, B., Pauchard, A., & Aguayo, M. (2010). *Pinus contorta* invasion in the Chilean Patagonia: Local patterns in a global context. *Biological Invasions, 12*(12), 3961–3971. https://doi.org/10.1007/s10530-010-9817-5

Langmaier, M., & Lapin, K. (2020). A systematic review of the impact of invasive alien plants on forest regeneration in European temperate forests. *Frontiers in Plant Science, 11*, 524969. https://doi.org/10.3389/fpls.2020.524969

Lankau, R., Jørgensen, P. S., Harris, D. J., & Sih, A. (2011). Incorporating evolutionary principles into environmental management and policy. *Evolutionary Applications, 4*(2), 315–325. https://doi.org/10.1111/j.1752-4571.2010.00171.x

Largiadèr, C. R. (2008). Hybridization and introgression between native and alien species. In W. Nentwig (Ed.), *Biological Invasions* (Vol. 193, pp. 275–292). Springer. https://doi.org/10.1007/978-3-540-36920-2_16

Larson, D. L., Anderson, P. J., & Newton, W. (2001). Alien plant invasion in mixed-grass prairie: Effects of vegetation type and anthropogenic disturbance. *Ecological Applications, 11*(1), 128–141. JSTOR. https://doi.org/10.2307/3061061

Latombe, G., Pyšek, P., Jeschke, J. M., Blackburn, T. M., Bacher, S., Capinha, C., Costello, M. J., Fernández, M., Gregory, R. D., Hobern, D., Hui, C., Jetz, W., Kumschick, S., McGrannachan, C., Pergl, J., Roy, H. E., Scalera, R., Squires, Z. E., Wilson, J. R. U., … McGeoch, M. A. (2017). A vision for global monitoring of biological invasions. *Biological Conservation, 213*, 295–308. https://doi.org/10.1016/j.biocon.2016.06.013

Lavoie, C., Saint-Louis, A., Guay, G., & Groeneveld, E. (2012). Les plantes vasculaires exotiques naturalisées: Une nouvelle liste pour le Québec. *Le Naturaliste canadien, 136*(3), 6–32. https://doi.org/10.7202/1009237ar

Lazkov, G. A., & Sultanova, B. A. (2011). *Checklist of vascular plants of Kyrgyzstan (in Russian)* (A. N. Sennikov, Ed.; Vol. 24). Finnish Museum of Natural History.

Leclerc, C., Courchamp, F., & Bellard, C. (2018). Insular threat associations within taxa worldwide. *Scientific Reports, 8*(1), 6393. https://doi.org/10.1038/s41598-018-24733-0

Leclerc, C., Villéger, S., Marino, C., & Bellard, C. (2020). Global changes threaten functional and taxonomic diversity of insular species worldwide. *Diversity and Distributions, 26*(4), 402–414. https://doi.org/10.1111/ddi.13024

Lecomte, F., Beall, E., Chat, J., Davaine, P., & Gaudin, P. (2013). The complete history of salmonid introductions in the Kerguelen Islands, Southern Ocean. *Polar Biology, 36*(4), 457–475. https://doi.org/10.1007/s00300-012-1281-5

Lee II, H., & Reusser, D. A. (2012). *Atlas of Nonindigenous Marine and Estuarine Species in the North Pacific*. Office of Research and Development, National Health and Environmental Effects Research Laboratory.

Lee, K.-H., Chen, T.-H., Shang, G., Clulow, S., Yang, Y.-J., & Lin, S.-M. (2019). A check list and population trends of invasive amphibians and reptiles in Taiwan. *ZooKeys, 829*, 85–130. https://doi.org/10.3897/zookeys.829.27535

Lehan, N. E., Murphy, J. R., Thorburn, L. P., & Bradley, B. A. (2013). Accidental introductions are an important source of invasive plants in the continental United States. *American Journal of Botany, 100*(7), 1287–1293. https://doi.org/10.3732/ajb.1300061

Leignel, V., Stillman, J. H., Baringou, S., Thabet, R., & Metais, I. (2014). Overview on the European green crab *Carcinus* spp. (Portunidae, Decapoda), one of the most famous marine invaders and ecotoxicological models. *Environmental Science and Pollution Research, 21*(15), 9129–9144. https://doi.org/10.1007/s11356-014-2979-4

LeJeune, K. D., & Seastedt, T. R. (2001). *Centaurea* species: The forb that won the west. *Conservation Biology, 15*(6), 1568–1574. JSTOR. http://www.jstor.org/stable/3061258

Lembrechts, J. J., Alexander, J. M., Cavieres, L. A., Haider, S., Lenoir, J., Kueffer, C., McDougall, K., Naylor, B. J., Nuñez, M. A., Pauchard, A., Rew, L. J., Nijs, I., & Milbau, A. (2017). Mountain roads shift native and non-native plant species' ranges. *Ecography, 40*(3), 353–364. https://doi.org/10.1111/ecog.02200

Lenda, M., Skórka, P., Knops, J. M. H., Moroń, D., Sutherland, W. J., Kuszewska, K., & Woyciechowski, M. (2014). Effect of the Internet Commerce on Dispersal Modes of Invasive Alien Species. *PLoS ONE, 9*(6), e99786. https://doi.org/10.1371/journal.pone.0099786

Lenoir, J., & Svenning, J.-C. (2015). Climate-related range shifts – a global multidimensional synthesis and new research directions. *Ecography, 38*(1), 15–28. https://doi.org/10.1111/ecog.00967

Lento, J., Goedkoop, W., Culp, J., Christoffersen, K. S., Lárusson, K. F., Fefilova, E., Guðbergsson, G., Liljaniemi, P., Ólafsson, J. S., Sandøy, S., Zimmerman, C., Christensen, T., Chambers, P., Heino, J., Hellsten, S., Kahlert, M., Keck, F., Laske, S., Lau, D. C. P., … Svenning, M. (2019). *State of the Arctic Freshwater Biodiversity Report.* Conservation of Arctic Flora and Fauna International Secretariat. https://www.caff.is/freshwater/freshwater-monitoring-publications/488-state-of-the-arctic-freshwater-biodiversity-report-full-report

Lenz, M.-I., Galvin, S., Keppel, G., Gopaul, S., Kowasch, M., Dyer, M. J., Watling, D., Lodhar, S., Hanson, G. C., Erasmi, S., & Boehmer, H. J. (2021). Where to invade next: Inaction on biological invasions threatens sustainability in a small island developing state of the tropical South Pacific. In P. S. Low (Ed.), *Sustainable Development: Asia-Pacific Perspectives.* Cambridge University Press. https://repository.usp.ac.fj/12527/

Lenzner, B., Latombe, G., Capinha, C., Bellard, C., Courchamp, F., Diagne, C., Dullinger, S., Golivets, M., Irl, S. D. H., Kühn, I., Leung, B., Liu, C., Moser, D., Roura-Pascual, N., Seebens, H., Turbelin, A., Weigelt, P., & Essl, F. (2020). What will the future bring for biological invasions on islands? An expert-based assessment. *Frontiers in Ecology and Evolution, 8*, 280. https://doi.org/10.3389/fevo.2020.00280

Leostrin, A., & Pergl, J. (2021). Alien flora in a boreal region of European Russia: An example of Kostroma oblast. *Biological Invasions, 23*, 3337–3350. https://doi.org/10.1007/s10530-021-02589-2

Leppäkoski, E., Gollasch, S., & Olenin, S. (Eds.). (2002). *Invasive Aquatic Species of Europe. Distribution, Impacts and Management.* Springer Netherlands. https://doi.org/10.1007/978-94-015-9956-6

Leprieur, F., Beauchard, O., Blanchet, S., Oberdorff, T., & Brosse, S. (2008). Fish invasions in the world's river systems: When natural processes are blurred by human activities. *PLoS Biology, 6*(2), e28. https://doi.org/10.1371/journal.pbio.0060028

Lester, P. J. (2005). Determinants for the successful establishment of exotic ants in New Zealand. *Diversity and Distributions, 11*(4), 279–288. https://doi.org/10.1111/j.1366-9516.2005.00169.x

Lester, P. J., & Beggs, J. R. (2019). Invasion Success and Management Strategies for Social Vespula Wasps. *Annual Review of Entomology, 64*, 51–71. https://doi.org/10.1146/annurev-ento-011118-111812

Leung, B., Springborn, M. R., Turner, J. A., & Brockerhoff, E. G. (2014). Pathway-level risk analysis: The net present value of an invasive species policy in the US. *Frontiers in Ecology and the Environment, 12*(5), 273–279. https://doi.org/10.1890/130311

Leuven, R. S. E. W., van der Velde, G., Baijens, I., Snijders, J., van der Zwart, C., Lenders, H. J. R., & bij de Vaate, A. (2009). The river Rhine: A global highway for dispersal of aquatic invasive species. *Biological Invasions, 11*(9), 1989–2008. https://doi.org/10.1007/s10530-009-9491-7

Lever, C. (2003). *Naturalized reptiles and amphibians of the world.* Oxford University Press.

Levine, J. M., & D'Antonio, C. M. (2003). Forecasting Biological Invasions with Increasing International Trade. *Conservation Biology, 17*(1), 322–326. https://doi.org/10.1046/j.1523-1739.2003.02038.x

Lewis, J. S., Farnsworth, M. L., Burdett, C. L., Theobald, D. M., Gray, M., & Miller, R. S. (2017). Biotic and abiotic factors predicting the global distribution and population density of an invasive large mammal. *Scientific Reports, 7*, 44152. https://doi.org/10.1038/srep44152

Li, B., Liao, C., Zhang, X., Chen, H., Wang, Q., Chen, Z., Gan, X., Wu, J., Zhao, B., Ma, Z., Cheng, X., Jiang, L., & Chen, J. (2009). *Spartina alterniflora* invasions in the Yangtze River estuary, China: An overview of current status and ecosystem effects. *Wetland Restoration and Ecological Engineering, 35*(4), 511–520. https://doi.org/10.1016/j.ecoleng.2008.05.013

Li, H., Huang, X., & Zhan, A. (2020). Stress memory of recurrent environmental challenges in marine invasive species: *Ciona robusta* as a case study. *Frontiers in Physiology, 11*, 94. https://doi.org/10.3389/fphys.2020.00094

Li, H., Xiao, H., Peng, H., Han, H., & Xue, D. (2006). Potential global range expansion of a new invasive species, the erythrina gall wasp, *Quadrastichus erythrinae* Kim (Insecta: Hymenoptera: Eulophidae). *Raffles Bulletin of Zoology, 54*(2), 229–234. https://lkcnhm.nus.edu.sg/wp-content/uploads/sites/10/app/uploads/2017/06/54rbz229-234.pdf

Li, K., Vandermeer, J. H., & Perfecto, I. (2016). Disentangling endogenous versus exogenous pattern formation in spatial ecology: A case study of the ant *Azteca sericeasur* in southern Mexico. *Royal Society Open Science, 3*(5). https://doi.org/10.1098/rsos.160073

Li, X., Bellard, C., Hu, F., & Li, H. (2020). Effect of distance, area, and climate on the frequency of introduction and extinction events on islands and archipelagos. *Ecosphere, 11*(1), e03008. https://doi.org/10.1002/ecs2.3008

Li, X., Liu, X., Kraus, F., Tingley, R., & Li, Y. (2016). Risk of biological invasions is concentrated in biodiversity hotspots. *Frontiers in Ecology and the Environment, 14*(8), 411–417. https://doi.org/10.1002/fee.1321

Liao, I. C., & Chien, Y.-H. (2011). The Pacific white shrimp, *Litopenaeus vannamei*, in Asia: The world's most widely cultured alien crustacean. In B. S. Galil, P. Clark, & J. Carlton (Eds.), *In the wrong place—Alien marine crustaceans: Distribution, biology and impacts* (Vol. 6, pp. 489–519). Springer. https://doi.org/10.1007/978-94-007-0591-3_17

Licata, F., Ficetola, G. F., Freeman, K., Mahasoa, R. H., Ravololonarivo, V., Solofo Niaina Fidy, J. F., Koto-Jean, A. B., Nahavitatsara, E. R., Andreone, F., & Crottini, A. (2019). Abundance, distribution and spread of the invasive Asian toad *Duttaphrynus melanostictus* in eastern Madagascar. *Biological Invasions, 21*(5), 1615–1626. https://doi.org/10.1007/s10530-019-01920-2

Liebhold, A. M., Brockerhof, E. G., Garrett, L. J., Parke, J. L., & Britton, K. O. (2012). Live plant imports: The major pathway for forest insect and pathogen invasions of the US. *Frontiers in Ecology and Evolution, 10*(3), 135–143. https://doi.org/10.1890/110198

Liebhold, A. M., Brockerhoff, E. G., Kalisz, S., Nuñez, M. A., Wardle, D. A., & Wingfield, M. J. (2017). Biological invasions in forest ecosystems. *Biological Invasions, 19*(11), 3437–3458. https://doi.org/10.1007/s10530-017-1458-5

Liebhold, A. M., & Griffin, R. L. (2016). The legacy of Charles Marlatt and efforts to limit plant pest invasions. *American Entomologist, 62*(4), 218–227. https://doi.org/10.1093/ae/tmw072

Liebhold, A. M., Yamanaka, T., Roques, A., Augustin, S., Chown, S. L., Brockerhoff, E. G., & Pyšek, P. (2018). Plant diversity drives global patterns of insect invasions. *Scientific Reports, 8*(1), 12095. https://doi.org/10.1038/s41598-018-30605-4

Liedtke, R., Barros, A., Essl, F., Lembrechts, J. J., Wedegärtner, R. E. M., Pauchard, A., & Dullinger, S. (2020). Hiking trails as conduits for the spread of non-native species in mountain areas. *Biological Invasions, 22*(3), 1121–1134. https://doi.org/10.1007/s10530-019-02165-9

Lin, S., Wang, J., LV, Z., Liang, P., Luo, L., Wang, X., & Wang, Y. (2017). First Record of An Alien Invasive Species *Eleutherodactylus planirostris* in Mainland China, and Its Population Study. *Sichuan Journal of Zoology, 36*(6), 680–685. https://doi.org/10.11984/j.issn.1000-7083.20170232

Lin, X., & Heitman, J. (2006). The Biology of the *Cryptococcus neoformans* Species Complex. *Annual Review of Microbiology, 60*(1), 69–105. https://doi.org/10.1146/annurev.micro.60.080805.142102

Lin, Y., Gao, Z., & Zhan, A. (2015). Introduction and use of non-native species for aquaculture in China: Status, risks and management solutions. *Reviews in Aquaculture, 7*(1), 28–58. https://doi.org/10.1111/raq.12052

Ling, S. D., Johnson, C. R., Ridgway, K., Hobday, A. J., & Haddon, M. (2009). Climate-driven range extension of a sea urchin: Inferring future trends by analysis of recent population dynamics. *Global Change Biology, 15*(3), 719–731. https://doi.org/10.1111/j.1365-2486.2008.01734.x

Lins, D. M., de Marco, P., Andrade, A. F. A., & Rocha, R. M. (2018). Predicting global ascidian invasions. *Diversity and Distributions, 24*(5), 692–704. https://doi.org/10.1111/ddi.12711

Linz, G. M., Homan, H. J., Gaulker, S. M., Penry, L. B., & Bleier, W. J. (2007). European starlings: A review of an invasive species with far-reaching impacts. *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, 24. https://digitalcommons.unl.edu/nwrcinvasive/24

Litchman, E. (2010). Invisible invaders: Non-pathogenic invasive microbes in aquatic and terrestrial ecosystems. *Ecology Letters, 13*(12), 1560–1572. https://doi.org/10.1111/j.1461-0248.2010.01544.x

Liu, C., Wolter, C., Xian, W., & Jeschke, J. M. (2020). Most invasive species largely conserve their climatic niche. *Proceedings of the National Academy of Sciences, 117*(38), 23643–23651. https://doi.org/10.1073/pnas.2004289117

Liu, J., & Li, Z. (2010). The Role of Exotics in Chinese Inland Aquaculture. In S. S. De Silva & F. B. Davy (Eds.), *Success Stories in Asian Aquaculture* (pp. 173–185). Springer. https://doi.org/10.1007/978-90-481-3087-0_9

Liu, X., Blackburn, T. M., Song, T., Li, X., Huang, C., & Li, Y. (2019). Risks of biological invasion on the belt and road. *Current Biology, 29*(3), 499-505.e4. https://doi.org/10.1016/j.cub.2018.12.036

Liu, X., Blackburn, T. M., Song, T., Wang, X., Huang, C., & Li, Y. (2020). Animal invaders threaten protected areas worldwide. *Nature Communications, 11*(1), 2892. https://doi.org/10.1038/s41467-020-16719-2

Liu, X., Li, X., Liu, Z., Tingley, R., Kraus, F., Guo, Z., & Li, Y. (2014). Congener diversity, topographic heterogeneity and human-assisted dispersal predict spread rates of alien herpetofauna at a global scale. *Ecology Letters, 17*(7), 821–829. https://doi.org/10.1111/ele.12286

Liu, X., Luo, Y., Chen, J. X., Guo, Y. S., Bai, C. M., & Li, Y. M. (2015). Diet and prey selection of the Invasive American bullfrog *(Lithobates catesbeianus)* in southwestern China. Asian Herpetological Research, 6(1), 34–44. https://doi.org/10.16373/j.cnki.ahr.140044

Liu, X., McGarrity, M. E., & Li, Y. (2012). The influence of traditional Buddhist wildlife release on biological invasions. *Conservation Letters, 5*(2), 107–114. https://doi.org/10.1111/j.1755-263X.2011.00215.x

Lockwood, J. L., Welbourne, D. J., Romagosa, C. M., Cassey, P., Mandrak, N. E., Strecker, A., Leung, B., Stringham, O. C., Udell, B., Episcopio-Sturgeon, D. J., Tlusty, M. F., Sinclair, J., Springborn, M. R., Pienaar, E. F., Rhyne, A. L., & Keller, R. (2019). When pets become pests: The role of the exotic pet trade in producing invasive vertebrate animals. *Frontiers in Ecology and the Environment, 17*(6), 323–330. https://doi.org/10.1002/fee.2059

Lodge, D. M., Deines, A., Gherardi, F., Yeo, D. C. J., Arcella, T., Baldridge, A. K., Barnes, M. A., Chadderton, W. L., Feder, J. L., Gantz, C. A., Howard, G. W., Jerde, C. L., Peters, B. W., Peters, J. A., Sargent, L. W., Turner, C. R., Wittmann, M. E., & Zeng, Y. (2012). Global introductions of crayfishes: Evaluating the impact of species invasions on ecosystem services. *Annual Review of Ecology, Evolution, and Systematics, 43*(1), 449–472. https://doi.org/10.1146/annurev-ecolsys-111511-103919

Lofgren, L. A., & Stajich, J. E. (2021). Fungal biodiversity and conservation mycology in light of new technology, big data, and changing attitudes. *Current Biology, 31*(19), R1312–R1325. https://doi.org/10.1016/j.cub.2021.06.083

Long, J. L. (2003). *Introduced Mammals of the World: Their History, Distribution and Influence.* CABI Publishing. https://doi.org/10.1071/9780643090156

Longcore, J. E., Pessier, A. P., & Nichols, D. K. (1999). *Batrachochytrium dendrobatidis* gen. Et sp. Nov., a chytrid pathogenic to amphibians. *Mycologia, 91*(2), 219–227. https://doi.org/10.2307/3761366

Lonsdale, W. M. (1994). Inviting trouble: Introduced pasture species in northern Australia. *Australian Journal of Ecology, 19*(3), 345–354. https://doi.org/10.1111/j.1442-9993.1994.tb00498.x

Loope, L. L., Nagata, R. J., & Medeiros, A. C. (1992). Alien plants in Haleakala National Park. In C. P. Stone, C. W. Smith, & J. T. Tunison (Eds.), *Alien plant invasions in native ecosystems of Hawai'i* (pp. 551–576). University of Hawaii Cooperative National Park Resources Studies Unit.

López-Gómez, M. J., Aguilar-Perera, A., & Perera-Chan, L. (2014). Mayan diver-fishers as citizen scientists: Detection and monitoring of the invasive red lionfish in the Parque Nacional Arrecife Alacranes, southern Gulf of Mexico. *Biological Invasions, 16*(7), 1351–1357. https://doi.org/10.1007/s10530-013-0582-0

Lososová, Z., Tichý, L., Divíšek, J., Čeplová, N., Danihelka, J., Dřevojan, P., Fajmon, K., Kalníková, V., Kalusová, V., Novák, P., Řehořek, V., Wirth, T., & Chytrý, M. (2018). Projecting potential future shifts in species composition of European urban plant communities. *Diversity and Distributions, 24*(6), 765–775. https://doi.org/10.1111/ddi.12725

Loss, S. R., & Marra, P. P. (2017). Population impacts of free-ranging domestic cats on mainland vertebrates. *Frontiers in Ecology and the Environment, 15*(9), 502–509. https://doi.org/10.1002/fee.1633

Louppe, V., Leroy, B., Herrel, A., & Veron, G. (2019). Current and future climatic regions favourable for a globally introduced wild carnivore, the raccoon *Procyon lotor. Scientific Reports, 9*, 9174. https://doi.org/10.1038/s41598-019-45713-y

Louppe, V., Leroy, B., Herrel, A., & Veron, G. (2020). The globally invasive small Indian mongoose *Urva auropunctata* is likely to spread with climate change. *Scientific Reports, 10*(1), 7461. https://doi.org/10.1038/s41598-020-64502-6

Lowry, B. J., Lowry, J. H., Jarvis, K. J., Keppel, G., Thaman, R. R., & Boehmer, H. J. (2020). Spatial patterns of presence, abundance, and richness of invasive woody plants in relation to urbanization in a tropical island setting. *Urban Forestry & Urban Greening, 48*, 126516. https://doi.org/10.1016/j.ufug.2019.126516

Lu, Y., Zhao, Q., Cheng, L., Zhao, L., Zhang, H., & Wei, J. (2020). The potential global distribution of the white peach scale *Pseudaulacaspis pentagona* (Targioni Tozzetti) under climate change. *Forest, 11*(2), 192. https://doi.org/10.3390/f11020192

Lubchenco, J. (1978). Plant species diversity in a marine intertidal community: Importance of herbivore food preference and algal competitive abilities. *The American Naturalist, 112*(983), 23–39. https://www.jstor.org/stable/2460135

Luja, V. H., Rodríguez-Estrella, R., Ratzlaff, K., Parra-Olea, G., & Ramírez-Bautista, A. (2012). The chytrid fungus *Batrachochytrium dendrobatidis* in isolated populations of the Baja California treefrog *Pseudacris hypochondriaca curta* in Baja California Sur, Mexico. *The Southwestern Naturalist, 57*(3), 323–327. https://doi.org/10.1894/0038-4909-57.3.323

Luken, J. O., & Thieret, J. W. (Eds.). (1997). *Assessment and management of plant invasions.* Springer New York. https://doi.org/10.1007/978-1-4612-1926-2

Luo, D., Wei, H., Chaichana, R., Yang, D., Gu, D., Mu, X., Xu, M., Yang, Y., Jin, S., & Hu, Y. (2019). Current status and potential risks of established alien fish species in China. *Aquatic Ecosystem Health & Management, 22*(4), 371–384. https://doi.org/10.1080/14634988.2019.1693226

Lutaenko, K. A., Furota, T., Nakayama, S., Shin, K., & Xu, J. (2013). *Atlas of Marine Invasive Species in the NOWPAP Region.* NOWPAP DINRAC (Northwest Pacific Action Plan, Data and Information Network Regional Center). https://wedocs.unep.org/20.500.11822/26336

Ma, X., Bangxi, X., Yindong, W., & Mingxue, W. (2003). Intentionally Introduced and Transferred Fishes in China's Inland Waters. *Asian Fisheries Science, 16*(4), 279–290. https://doi.org/10.33997/j.afs.2003.16.4.001

Macaya, E. C., López, B., Tala, F., Tellier, F., & Thiel, M. (2016). Float and raft: The role of buoyant seaweeds in the phylogeography and genetic structure of non-buoyant associated flora. In Z.-M. Hu & C. Fraser (Eds.), *Seaweed Phylogeography: Adaptation and Evolution of Seaweeds under Environmental Change* (pp. 97–130). Springer Netherlands. https://doi.org/10.1007/978-94-017-7534-2_4

Macdonald, I. A. W. (1983). Alien trees, shrubs and creepers invading indigenous vegetation in the Hluhluwe-Umfolozi Game Reserve Complex in Natal. *Bothalia, 14*(3/4), 949–959. https://doi.org/10.4102/abc.v14i3/4.1268

Mack, R. N. (1989). Temperate grasslands vulnerable to plant invasions: Characteristics and consequences. In J. A. Drake, H. A. Mooney, F. di Castri, R. H. Groves, F. J. Kruger, M. Rejmanek, & M. H. Williamson (Eds.), *Biological invasions: A global perspective* (pp. 155–179). Wiley. https://www.worldcat.org/title/biological-invasions-a-global-perspective/oclc/18323459

Mack, R. N., & Erneberg, M. (2002). The United States naturalized flora: Largely the product of deliberate introductions. *Annals of the Missouri Botanical Garden, 89*(2), 176–189. https://doi.org/10.2307/3298562

Madzivanzira, T. C., South, J., Wood, L. E., Nunes, A. L., & Weyl, O. L. F. (2021). A Review of Freshwater Crayfish Introductions in Africa. *Reviews in Fisheries Science & Aquaculture, 29*(2), 218–241. https://doi.org/10.1080/23308249.2020.1802405

Magalhães, A. L. B., Daga, V. S., Bezerra, L. A. V., Vitule, J. R. S., Jacobi, C. M., & Silva, L. G. M. (2020). All the colors of the world: Biotic homogenization-differentiation dynamics of freshwater fish communities on demand of the Brazilian aquarium trade. *Hydrobiologia, 847*(18), 3897–3915. https://doi.org/10.1007/s10750-020-04307-w

Magory Cohen, T., McKinney, M., Kark, S., & Dor, R. (2019). Global invasion in progress: Modeling the past, current and potential global distribution of the common myna. *Biological Invasions, 21*(4), 1295–1309. https://doi.org/10.1007/s10530-018-1900-3

Mahoney, P. J., Beard, K. H., Durso, A. M., Tallian, A. G., Long, A. L., Kindermann, R. J., Nolan, N. E., Kinka, D., & Mohn, H. E. (2014). Introduction effort, climate matching and species traits as predictors of global establishment success in non-native reptiles. *Diversity and Distributions, 21*, 64–74. https://doi.org/10.1111/ddi.12240

Máiz-Tomé, L., Sayer, C., & Darwall, W. R. T. (Eds.). (2018). *The status and distribution of freshwater biodiversity in Madagascar and the Indian Ocean islands hotspot.* IUCN. https://doi.org/10.2305/IUCN.CH.2018.RA.1.en

Major, K. C., Nosko, P., Kuehne, C., Campbell, D., & Bauhus, J. (2013). Regeneration dynamics of non-native northern red oak (*Quercus rubra* L.) populations as influenced by environmental factors: A case study in managed hardwood forests of southwestern Germany. *Forest Ecology and Management, 291*, 144–153. https://doi.org/10.1016/j.foreco.2012.12.006

Mangiante, M. J., Davis, A. J. S., Panlasigui, S., Neilson, M. E., Pfingsten, I., Fuller, P. L., & Darling, J. A. (2018). Trends in nonindigenous aquatic species richness in the United States reveal shifting spatial and temporal patterns of species introductions. *Aquatic Invasions, 13*(3), 323–338. https://doi.org/10.3391/ai.2018.13.3.02

Marchini, A., & Cardeccia, A. (2017). Alien amphipods in a sea of troubles: Cryptogenic species, unresolved taxonomy and overlooked introductions. *Marine Biology, 164*(4), 69. https://doi.org/10.1007/s00227-017-3093-1

Marchini, A., Galil, B. S., & Occhipinti-Ambrogi, A. (2015). Recommendations on standardizing lists of marine alien species: Lessons from the Mediterranean Sea. *Marine Pollution Bulletin, 101*(1), 267–273. https://doi.org/10.1016/j.marpolbul.2015.09.054

Marchioro, C. A. (2016). Global Potential Distribution of *Bactrocera carambolae* and the Risks for Fruit Production in Brazil. *PLoS ONE, 11*(11), e0166142. https://doi.org/10.1371/journal.pone.0166142

Marchioro, C. A., & Krechemer, F. S. (2018). Potential global distribution of *Diabrotica* species and the risks for agricultural production. *Pest Management Science, 74*(9), 2100–2109. https://doi.org/10.1002/ps.4906

Margaritopoulos, J. T., Kasprowicz, L., Malloch, G. L., & Fenton, B. (2009). Tracking the global dispersal of a cosmopolitan insect pest, the peach potato aphid. *BMC Ecology, 9*(1), 13. https://doi.org/10.1186/1472-6785-9-13

Marichal, R., Grimaldi, M., Mathieu, J., Brown, G. G., Desjardins, T., Silva Junior, M. L. da, Praxedes, C., Martins, M. B., Velasquez, E., & Lavelle, P. (2012). Is invasion of deforested Amazonia by the earthworm *Pontoscolex corethrurus* driven by soil texture and chemical properties? *Pedobiologia, 55*(5), 233–240. https://doi.org/10.1016/j.pedobi.2012.03.006

Marini, L., Bertolli, A., Bona, E., Federici, G., Martini, F., Prosser, F., & Bommarco, R. (2013). Beta-diversity patterns elucidate mechanisms of alien plant invasion in mountains. *Global Ecology and Biogeography, 22*(4), 450–460. https://doi.org/10.1111/geb.12006

Marler, T. E. (2020). Three invasive tree species change soil chemistry in Guam forests. *Forests, 11*(3), 279. https://doi.org/10.3390/f11030279

Maroyi, A. (2012). The casual, naturalised and invasive alien flora of Zimbabwe based on herbarium and literature records. *Koedoe, 54*(1), a1054. https://doi.org/10.4102/koedoe.v54i1.1054

Marr, S. M., Olden, J. D., Leprieur, F., Arismendi, I., Ćaleta, M., Morgan, D. L., Nocita, A., Šanda, R., Serhan Tarkan, A., & García-Berthou, E. (2013). A global assessment of freshwater fish introductions in mediterranean-climate regions. *Hydrobiologia, 719*(1), 317–329. https://doi.org/10.1007/s10750-013-1486-9

Marshall, B. M., Casewell, N. R., Vences, M., Glaw, F., Andreone, F., Rakotoarison, A., Zancolli, G., Woog, F., & Wüster, W. (2018). Widespread vulnerability of Malagasy predators to the toxins of an introduced toad. *Current Biology, 28*(11), R654–R655.

Marshall, V. M., Lewis, M. M., & Ostendorf, B. (2012). Buffel grass *(Cenchrus ciliaris)* as an invader and threat to biodiversity in arid environments: A review. *Journal of Arid Environments, 78*, 1–12. https://doi.org/10.1016/j.jaridenv.2011.11.005

Martin, L. J., Blossey, B., & Ellis, E. (2012). Mapping where ecologists work: Biases in the global distribution of terrestrial ecological observations. *Frontiers in Ecology and the Environment, 10*(4), 195–201. https://doi.org/10.1890/110154

Martin, P. H., Canham, C. D., & Marks, P. L. (2009). Why forests appear resistant to exotic plant invasions: Intentional introductions, stand dynamics, and the role of shade tolerance. *Frontiers in Ecology and the Environment, 7*(3), 142–149. https://doi.org/10.1890/070096

Martinez, J., Melo, T., Wong, L. J., & Pagad, S. (2021). *Global Register of Introduced and Invasive Species—Cabo Verde* [dataset]. Invasive Species Specialist Group ISSG. https://doi.org/10.15468/XBODNZ

Martins, D. J., Miller, S. E., Cords, M., Hirschauer, M. T., & Goodale, C. B. (2015). Observations on an Irruption Event of the Moth *Achaea catocaloides* (Lepidoptera: Erebidae) at Kakamega Forest, Kenya. *Journal of East African Natural History, 103*(1), 31–38. https://doi.org/10.2982/028.103.0102

Masin, S., Bonardi, A., Padoa-Schioppa, E., Bottoni, L., & Ficetola, G. F. (2014). Risk of invasion by frequently traded freshwater turtles. *Biological Invasions, 16*, 217–231. https://doi.org/10.1007/s10530-013-0515-y

Maskell, L. C., Bullock, J. M., Smart, S. M., Thompson, K., & Hulme, P. E. (2006). The distribution and habitat associations of non-native plant species in urban riparian habitats. *Journal of Vegetation Science, 17*(4), 499–508. https://doi.org/10.1111/j.1654-1103.2006.tb02471.x

Masocha, M., & Dube, T. (2018). Global terrestrial biomes at risk of cacti invasion identified for four species using consensual modelling. *Journal of Arid Environments, 156*, 77–86. https://doi.org/10.1016/j.jaridenv.2018.05.006

Masseti, M. (2009). Mammals of the Mediterranean islands: Homogenisation and the loss of biodiversity. *Mammalia, 73*(3), 169–202. https://doi.org/10.1515/MAMM.2009.029

Masseti, M., & Zuffi, M. A. L. (2011). On the origin of the asp viper *Vipera aspis hugyi* Schinz, 1833, on the island of Montecristo, Northern Tyrrhenian Sea (Tuscan archipelago, Italy). *The Herpetological Bulletin, 117*, 1–9. https://www.researchgate.net/publication/221935738_On_the_origin_of_the_asp_viper_Vipera_aspis_hugyi_Schinz_1833_on_the_island_of_Montecristo_Northern_Tyrrhenian_Sea_Tuscan_archipelago_Italy

Mateo, J. A., Ayres, C., & Lopez-Jurado, L. F. (2011). Los anfibios y reptiles naturalizados en España. Historia y evolución de una problemática creciente. *Boletín de La Asociación Herpetológica Española, 22*, 2–42. http://hdl.handle.net/10553/18273

Mathieson, A. C., Dawes, C. J., Harris, L. G., & Hehre, E. J. (2003). Expansion of the Asiatic green alga *Codium Fragile* subsp. *Tomentosoides* in the Gulf of Maine. *Rhodora, 105*(921), 1–53. https://www.jstor.org/stable/23313587

Mattson, W. J., Niemela, P., Millers, I., & Inguanzo, Y. (1994). *Immigrant Phytophagous Insects on Woody Plants in the United States and Canada: An Annotated List.* US Department of Agriculture. https://doi.org/10.2737/NC-GTR-169

Mattson, W. J., Vanhanen, H., Veteli, T., Sivonen, S., & Niemelä, P. (2007). Few immigrant phytophagous insects on woody plants in Europe: Legacy of the European crucible? *Biological Invasions, 9*(8), 957–974. https://doi.org/10.1007/s10530-007-9096-y

Mawarda, P. C., Le Roux, X., Dirk van Elsas, J., & Salles, J. F. (2020). Deliberate introduction of invisible invaders: A critical appraisal of the impact of microbial inoculants on soil microbial communities. *Soil Biology and Biochemistry, 148*. https://doi.org/10.1016/j.soilbio.2020.107874

Mazaris, A. D., & Katsanevakis, S. (2018). The threat of biological invasions is under-represented in the marine protected areas of the European Natura 2000 network. *Biological Conservation, 225*, 208–212. https://doi.org/10.1016/j.biocon.2018.07.007

Mazía, C. N., Chaneton, E. J., Machera, M., Uchitel, A., Feler, M. V., & Ghersa, C. M. (2010). Antagonistic effects of large- and small-scale disturbances on exotic tree invasion in a native tussock grassland relict. *Biological Invasions, 12*(9), 3109–3122. https://doi.org/10.1007/s10530-010-9702-2

McCarthy, A. H., Peck, L. S., Hughes, K. A., & Aldridge, D. C. (2019). Antarctica: The final frontier for marine biological invasions. *Global Change Biology, 25*(7), 2221–2241. https://doi.org/10.1111/gcb.14600

McCartney, M. A., Mallez, S., & Gohl, D. M. (2019). Genome projects in invasion biology. *Conservation Genetics, 20*(6), 1201–1222. https://doi.org/10.1007/s10592-019-01224-x

McCay, T. S., & Scull, P. (2019). Invasive lumbricid earthworms in northeastern North American forests and consequences for leaf-litter fauna. *Biological Invasions, 21*(6), 2081–2093. https://doi.org/10.1007/s10530-019-01959-1

McClain, C. R., Allen, A. P., Tittensor, D. P., & Rex, M. A. (2012). Energetics of life on the deep seafloor. *Proceedings of the National Academy of Sciences, 109*(38), 15366–15371. https://doi.org/10.1073/pnas.1208976109

McDougall, K. L., Alexander, J. M., Haider, S., Pauchard, A., Walsh, N. G., & Kueffer, C. (2011). Alien flora of mountains: Global comparisons for the development of local preventive measures against plant invasions. *Diversity and Distributions, 17*(1), 103–111. https://doi.org/10.1111/j.1472-4642.2010.00713.x

McDougall, K. L., Lembrechts, J., Rew, L. J., Haider, S., Cavieres, L. A., Kueffer, C., Milbau, A., Naylor, B. J., Nuñez, M. A., Pauchard, A., Seipel, T., Speziale, K. L., Wright, G. T., & Alexander, J. M. (2018). Running off the road: Roadside non-native plants invading mountain vegetation. *Biological Invasions, 20*(12), 3461–3473. https://doi.org/10.1007/s10530-018-1787-z

McDougall, K. L., Morgan, J. W., Walsh, N. G., & Williams, R. J. (2005). Plant invasions in treeless vegetation of the Australian Alps. *Perspectives in Plant Ecology, Evolution and Systematics, 7*(3), 159–171. https://doi.org/10.1016/j.ppees.2005.09.001

McGeoch, M. A., Butchart, S. H. M., Spear, D., Marais, E., Kleynhans, E. J., Symes, A., Chanson, J., & Hoffmann, M. (2010). Global indicators of biological invasion: Species numbers, biodiversity impact and policy responses. *Diversity and Distributions, 16*(1), 95–108. https://doi.org/10.1111/j.1472-4642.2009.00633.x

McGeoch, M. A., Shaw, J. D., Terauds, A., Lee, J. E., & Chown, S. L. (2015). Monitoring biological invasion across the broader Antarctic: A baseline and indicator framework. *Global Environmental Change, 32*, 108–125. https://doi.org/10.1016/j.gloenvcha.2014.12.012

Mckindsey, C. W., Landry, T., O'beirn, F. X., & Davies, I. M. (2007). Bivalve aquaculture and exotic species: A review of ecological considerations and management issues. *Journal of Shellfish Research, 26*(2), 281–294. https://doi.org/10.2983/0730-8000(2007)26[281:BAAESA]2.0.CO;2

McKinley, D. C., Miller-Rushing, A. J., Ballard, H. L., Bonney, R., Brown, H., Cook-Patton, S. C., Evans, D. M., French, R. A., Parrish, J. K., Phillips, T. B., Ryan, S. F., Shanley, L. A., Shirk, J. L., Stepenuck, K. F., Weltzin, J. F., Wiggins, A., Boyle, O. D., Briggs, R. D., Chapin, S. F., … Soukup, M. A. (2017). Citizen science can improve conservation science, natural resource management, and environmental protection. *Biological Conservation, 208*, 15–28. https://doi.org/10.1016/j.biocon.2016.05.015

McLean, P., Gallien, L., Wilson, J. R. U., Gaertner, M., & Richardson, D. M. (2017). Small urban centres as launching sites for plant invasions in natural areas: Insights from South Africa. *Biological Invasions, 19*(12), 3541–3555. https://doi.org/10.1007/s10530-017-1600-4

McMullan, M., Rafiqi, M., Kaithakottil, G., Clavijo, B. J., Bilham, L., Orton, E., Percival-Alwyn, L., Ward, B. J., Edwards, A., Saunders, D. G. O., Garcia Accinelli, G., Wright, J., Verweij, W., Koutsovoulos, G., Yoshida, K., Hosoya, T., Williamson, L., Jennings, P., Ioos, R., … Clark, M. D. (2018). The ash dieback invasion of Europe was founded by two genetically divergent individuals. *Nature Ecology & Evolution, 2*(6), 1000–1008. https://doi.org/10.1038/s41559-018-0548-9

McTaggart, A. R., Shivas, R. G., van der Nest, M. A., Roux, J., Wingfield, B. D., & Wingfield, M. J. (2016). Host jumps shaped the diversity of extant rust fungi (Pucciniales). *New Phytologist, 209*(3), 1149–1158. https://doi.org/10.1111/nph.13686

Mead, A., Carlton, J. T., Griffiths, C. L., & Rius, M. (2011). Revealing the scale of marine bioinvasions in developing regions: A South African re-assessment. *Biological Invasions, 13*(9), 1991–2008. https://doi.org/10.1007/s10530-011-0016-9

Measey, J., Davies, S. J., Vimercati, G., Rebelo, A., Schmidt, W., & Turner, A. (2017). Invasive amphibians in southern Africa: A review of invasion pathways. *Bothalia-African Biodiversity & Conservation, 47*(2), 1–12. https://doi.org/10.4102/abc.v47i2.2117

Meddour, R., Sahar, O., & Fried, G. (2020). A preliminary checklist of the alien flora of Algeria (North Africa): Taxonomy, traits and invasiveness potential. *Botany Letters, 167*(4), 453–470. https://doi.org/10.1080/23818107.2020.1802775

Medina, A., Gomes, I., Araújo, S., Lima, L., & Monteiro, R. (2015). *Fifth National Report on The Status of Biodiversity in Cabo Verde* (National Directorate for The Environment). https://www.cbd.int/doc/world/cv/cv-nr-05-en.pdf

Meineke, E. K., Dunn, R. R., Sexton, J. O., & Frank, S. D. (2013). Urban Warming Drives Insect Pest Abundance on Street Trees. *PLoS ONE, 8*(3), e59687. https://doi.org/10.1371/journal.pone.0059687

Meiners, S. J., & Pickett, S. T. A. (2013). Plant Invasions in Protected Landscapes: Exception or Expectation? In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas* (Vol. 7, pp. 43–60). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_3

Mertelmeyer, L., Jacobi, J. D., Mueller-Dombois, D., Brinck, K., & Boehmer, H. J. (2019). Regeneration of *Metrosideros polymorpha* forests in Hawaii after landscape-level canopy dieback. *Journal of Vegetation Science, 30*(1), 146–155. https://doi.org/10.1111/jvs.12704

Meshaka, W. E., Jr. (2011). A runaway train in the making: The exotic amphibians, reptiles, turtles, and crocodilians of Florida. In *Herpetological Conservation and Biology* (Vol. 6, pp. 1–101).

Meurisse, N., Rassati, D., Hurley, B. P., Brockerhoff, E. G., & Haack, R. A. (2019). Common pathways by which non-native forest insects move internationally and domestically. *Journal of Pest Science, 92*(1), 13–27. https://doi.org/10.1007/s10340-018-0990-0

Meyer, C., Kreft, H., Guralnick, R., & Jetz, W. (2015). Global priorities for an effective information basis of biodiversity distributions. *Nature Communications, 6*(1), 8221. https://doi.org/10.1038/ncomms9221

Meyer, C., Weigelt, P., & Kreft, H. (2016). Multidimensional biases, gaps and uncertainties in global plant occurrence information. *Ecology Letters, 19*(8), 992–1006. https://doi.org/10.1111/ele.12624

Meyer, J.-Y. (2014). Critical issues and new challenges for research and management of invasive plants in the Pacific Islands. *Pacific Conservation Biology, 20*(2), 146–164. https://doi.org/10.1071/pc140146

Meyer, J.-Y., & Butaud, J.-F. (2009). The impacts of rats on the endangered native flora of French Polynesia (Pacific Islands): Drivers of plant extinction or *coup de grâce* species? *Biological Invasions, 11*(7), 1569–1585. https://doi.org/10.1007/s10530-008-9407-y

Meyer, J.-Y., & Florence, J. (1996). Tahiti's native flora endangered by the invasion of *Miconia calvescens* DC. (Melastomataceae). *Journal of Biogeography, 23*(6), 775–781. https://doi.org/10.1111/j.1365-2699.1996.tb00038.x

Meyer, J.-Y., Taureau, F., Bisarah, L., Madi Moussa, R., Gorchakova, E., & Caillaud, A. (2021). Introduced mangroves in the Society Islands, French Polynesia (South Pacific): Invasive species or novel ecosystem? *Biological Invasions, 23*(8), 2527–2539. https://doi.org/10.1007/s10530-021-02520-9

Meyer, W. M., Hayes, K. A., & Meyer, A. L. (2008). Giant African snail, *Achatina fulica*, as a snail predator. *American Malacological Bulletin, 24*(1), 117–119. https://doi.org/10.4003/0740-2783-24.1.117

Meyerson, L. A., & Cronin, J. T. (2013). Evidence for multiple introductions of *Phragmites australis* to North America: Detection of a new non-native haplotype. *Biological Invasions, 15*(12), 2605–2608. https://doi.org/10.1007/s10530-013-0491-2

Meyerson, L. A., & Mooney, H. A. (2007). Invasive alien species in an era of globalization. *Frontiers in Ecology and the Environment, 5*(4), 199–208. https://doi.org/10.1890/1540-9295(2007)5[199:IASIAE]2.0.CO;2

Meyerson, L. A., Pauchard, A., Brundu, G., Carlton, J. T., Hierro, J. L., Kueffer, C., Pandit, M. K., Pyšek, P., Richardson, D. M., & Packer, J. G. (2022). Moving Toward Global Strategies for Managing Invasive Alien Species. In D. R. Clements, M. K. Upadhyaya, S. Joshi, & A. Shrestha (Eds.), *Global Plant Invasions* (pp. 331–360). Springer International Publishing. https://doi.org/10.1007/978-3-030-89684-3_16

Meyerson, L. A., Pyšek, P., Lučanová, M., Wigginton, S., Tran, C., & Cronin, J. T. (2020). Plant genome size influences stress tolerance of invasive and native plants via plasticity. *Ecosphere, 11*(5), e03145. https://doi.org/10.1002/ecs2.3145

Meyerson, L. A., & Reaser, J. K. (2003). *The Ecological and Socio-Economic Impacts of Invasive Alien Species on Island Ecosystems: Report of an Experts Consultation* [Washington, DC: The Global Invasive Species Programme (GISP) on behalf of the Convention on Biological Diversity.]. https://www.cbd.int/doc/ref/ais-gisp-report-en.pdf

Meyerson, L. A., Viola, D. V., & Brown, R. N. (2010). Hybridization of invasive *Phragmites australis* with a native subspecies in North America. *Biological Invasions, 12*(1), 103–111. https://doi.org/10.1007/s10530-009-9434-3

Migge-Kleian, S., McLean, M. A., Maerz, J. C., & Heneghan, L. (2006). The influence of invasive earthworms on indigenous fauna in ecosystems previously uninhabited by earthworms. *Biological Invasions, 8*(6), 1275–1285. https://doi.org/10.1007/s10530-006-9021-9

Miletti, T. E., Carlyle, C. E., Picard, C. R., Mulac, K. M., Landaw, A., & Fraser, L. H. (2005). Hydrology, water chemistry, and vegetation characteristics of a tamarack bog in Bath Township, Ohio: Towards restoration and enhancement. *The Ohio Journal of Science, 105*(2), 21–30. https://kb.osu.edu/handle/1811/31895

Millennium Ecosystem Assessment. (2005). *Ecosystems and Human Well-Being: Opportunities and Challenges for Business and Industry* (Vol. 5). Island Press. https://www.unioviedo.es/ranadon/Ricardo_Anadon/docencia/DoctoradoEconomia/Millenium%20Eco%20Assesment%2005%20Oppor%20Business%20Industry.pdf

Miller, A. W., & Ruiz, G. M. (2014). Arctic shipping and marine invaders. *Nature Climate Change, 4*(6), 413–416. https://doi.org/10.1038/nclimate2244

Mills, E. L., Leach, J. H., Carlton, J. T., & Secor, C. L. (1993). Exotic Species in the Great Lakes: A History of Biotic Crises and Anthropogenic Introductions. *Journal of Great Lakes Research, 19*(1), 1–54. https://doi.org/10.1016/S0380-1330(93)71197-1

Minden, V., Jacobi, J. D., Porembski, S., & Boehmer, H. J. (2010). Effects of invasive alien kahili ginger *(Hedychium gardnerianum)* on native plant species regeneration in a Hawaiian rainforest.

Applied Vegetation Science, 13(1), 5–14. https://doi.org/10.1111/j.1654-109X.2009.01056.x

Mineur, F., Belsher, T., Johnson, M. P., Maggs, C. A., & Verlaque, M. (2007). Experimental assessment of oyster transfers as a vector for macroalgal introductions. *Biological Conservation, 137*(2), 237–247. https://doi.org/10.1016/j.biocon.2007.02.001

Miranda, R. J., Nunes, J. A. C. C., Creed, J. C., Barros, F., Macieira, R. M., Santos, R. G., Lima, G. V., Pontes, A. V. F., Silva, L. G. F. C., Cordeiro, R. T., Sampaio, C. L. S., Pinto, T. K., Malhado, A. C. M., Ladle, R., & Pereira, P. H. (2020). Brazil policy invites marine invasive species. *Science, 368*(6490), 481. https://doi.org/10.1126/science.abb7255

Miranda-Chumacero, G., Wallace, R., Calderón, H., Calderón, G., Willink, P., Guerrero, M., Siles, T. M., Lara, K., & Chuqui, D. (2012). Distribution of arapaima *(Arapaima gigas)* (Pisces: Arapaimatidae) in Bolivia: Implications in the control and management of a non-native population. *Bioinvasion Records, 1*(2), 129–138. https://doi.org/10.3391/bir.2012.1.2.09

Miravete, V., Roura-Pascual, N., Dunn, R. R., & Gómez, C. (2014). How many and which ant species are being accidentally moved around the world? *Biology Letters, 10*(8), 20140518. https://doi.org/10.1098/rsbl.2014.0518

Mitchell, J. E. (2000). *Rangeland resource trends in the United States: A technical document supporting the 2000 USDA Forest Service RPA Assessment* (Gen. Tech. Rep. RMRS-GTR-68). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-68

Mittelhauser, J. R., Barnes, P. W., & Barnes, T. G. (2011). The Effect of Herbicide on the Re-Establishment of Native Grasses in the Blackland Prairie. *Natural Areas Journal, 31*(3), 226–233. https://doi.org/10.3375/043.031.0305

Mittermeier, R. A., Turner, W. R., Larsen, F. W., Brooks, T. M., & Gascon, C. (2011). Global Biodiversity Conservation: The Critical Role of Hotspots. In F. E. Zachos & J. C. Habel (Eds.), *Biodiversity Hotspots* (pp. 3–22). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-20992-5_1

Mo, M. (2017). Asian Black-spined Toads *(Duttaphrynus melanostictus)* in Australia: An invasion worth avoiding. *Reptiles & Amphibians, 24*(3), 155–161. https://doi.org/10.17161/randa.v24i3.14197

Mohamed, K. I., Papes, M., Williams, R., Benz, B. W., & Peterson, A. T. (2006). Global invasive potential of 10 parasitic witchweeds and related Orobanchaceae. *AMBIO, 35*(6), 281–288. https://doi.org/10.1579/05-R-051R.1

Mohanty, N. P., Crottini, A., Garcia, R. A., & Measey, J. (2021). Non-native populations and global invasion potential of the Indian bullfrog *Hoplobatrachus tigerinus*: A synthesis for risk-analysis. *Biological Invasions, 23*(1), 69–81. https://doi.org/10.1007/s10530-020-02356-9

Mohanty, N. P., & Measey, J. (2019). The global pet trade in amphibians: Species traits, taxonomic bias, and future directions. *Biodiversity and Conservation, 28*(14), 3915–3923. https://doi.org/10.1007/s10531-019-01857-x

Monaco, A., Genovesi, P., & Middleton, A. (2016). *European Code of Conduct on Hunting and Invasive Alien Species* (p. 42). Council of Europe. https://easin.jrc.ec.europa.eu/easin/Document/EuropeanCodeofConduct/Publication_Code_of_conduct_Hunting_IAS_2016_Web.pdf

Monteiro, M., Reino, L., Schertler, A., Essl, F., Figueira, R., Ferreira, M. T., & Capinha, C. (2020). A database of the global distribution of alien macrofungi. *Biodiversity Data Journal, 8*, e51459. https://doi.org/10.3897/BDJ.8.e51459

Mooney, H. A., Mack, R., McNeely, J. A., Neville, L. E., Schei, P. J., & Waage, J. K. (Eds.). (2005). *Invasive Alien Species: A New Synthesis* (Vol. 63). Island Press. https://books.google.co.za/books?id=KBqYU2dC-b4C

Moore, M., Francois Solofo Niaina Fidy, J., & Edmonds, D. (2015). The New Toad in Town: Distribution of the Asian Toad, *Duttaphrynus Melanostictus*, in the Toamasina Area of Eastern Madagascar. *Tropical Conservation Science, 8*(2), 440–455. https://doi.org/10.1177/194008291500800210

Morales, C. L., Sáez, A., Garibaldi, L. A., & Aizen, M. A. (2017). Disruption of Pollination Services by Invasive Pollinator Species. In M. Vilà & P. E. Hulme (Eds.), *Impact of Biological Invasions on Ecosystem Services* (Vol. 12, pp. 203–220). Springer International Publishing. https://doi.org/10.1007/978-3-319-45121-3_13

Moran, E. V., & Alexander, J. M. (2014). Evolutionary responses to global change: Lessons from invasive species. *Ecology Letters, 17*(5), 637–649. https://doi.org/10.1111/ele.12262

Moreno, R. A., Neill, P. E., & Rozbaczylo, N. (2006). Native and non-indigenous boring polychaetes in Chile: A threat to native and commercial mollusc species. *Revista Chilena de Historia Natural, 79*(2), 263–278. https://doi.org/10.4067/S0716-078X2006000200012

Morimoto, N., Kiritani, K., Yamamura, K., & Yamanaka, T. (2019). Finding indications of lag time, saturation and trading inflow in the emergence record of exotic agricultural insect pests in Japan. *Applied Entomology and Zoology, 54*, 437–450. https://doi.org/10.1007/s13355-019-00640-2

Moritz, R. F. A., Härtel, S., & Neumann, P. (2005). Global invasions of the western honeybee (Apis mellifera) and the consequences for biodiversity. *Écoscience, 12*(3), 289–301. https://doi.org/10.2980/i1195-6860-12-3-289.1

Moser, D., Lenzner, B., Weigelt, P., Dawson, W., Kreft, H., Pergl, J., Pyšek, P., van Kleunen, M., Winter, M., Capinha, C., Cassey, P., Dullinger, S., Economo, E. P., García-Díaz, P., Guénard, B., Hofhansl, F., Mang, T., Seebens, H., & Essl, F. (2018). Remoteness promotes biological invasions on islands worldwide. *Proceedings of the National Academy of Sciences, 115*(37), 9270–9275. https://doi.org/10.1073/pnas.1804179115

Motiejūnaitė, J., Kutorga, E., Kasparavičius, J., Lygis, V., & Norkutė, G. (2016). New records from Lithuania of fungi alien to Europe. *Mycotaxon, 131*(1), 49–60. https://doi.org/10.5248/131.49

Moulton, M. P., & Pimm, S. L. (1983). The introduced Hawaiian avifauna: Biogeographic evidence for competition. *The American Naturalist, 121*(5), 669–690. http://www.jstor.org/stable/2460871

Moyle, P. B. (1986). Fish Introductions into North America: Patterns and Ecological Impact. In H. A. Mooney & J. A. Drake (Eds.), *Ecology of Biological Invasions of North America and Hawaii* (Vol. 58, pp. 27–43). Springer New York. https://doi.org/10.1007/978-1-4612-4988-7_2

MPSC. (2018). *Australian Priority Marine Pest List: Process and outcomes.* Marine Pest Sectoral Committee, Australian Bureau of Agricultural and Resource Economics and Sciences. https://www.marinepests.gov.au/sites/default/files/Documents/apmpl-process-outcomes.pdf

Mrugała, A., Kozubíková-Balcarová, E., Chucholl, C., Cabanillas Resino, S., Viljamaa-Dirks, S., Vukić, J., & Petrusek, A. (2015). Trade of ornamental crayfish in Europe as a possible introduction pathway for important crustacean diseases: Crayfish plague and white spot syndrome. *Biological Invasions, 17*(5), 1313–1326. https://doi.org/10.1007/s10530-014-0795-x

Muirhead, J. R., Minton, M. S., Miller, W. A., & Ruiz, G. M. (2015). Projected effects of the Panama Canal expansion on shipping traffic and biological invasions. *Diversity and Distributions, 21*(1), 75–87. https://doi.org/10.1111/ddi.12260

Mukaromah, L., & Imron, M. A. (2019). Invasive Plant Species in The Disturbed Forest of Batukahu Nature Reserve, Bali, Indonesia. *BIOTROPIA – The Southeast Asian Journal of Tropical Biology, 27*(1), 22–32. https://doi.org/10.11598/btb.2020.27.1.933

Mukul, S. A., Arfin Khan, M. A. S., & Uddin, M. B. (2020). Identifying threats from invasive alien species in Bangladesh. *Global Ecology and Conservation, 23*, e01196. https://doi.org/10.1016/j.gecco.2020.e01196

Mulder, C. P. H., & Spellman, K. V. (2019). Do longer growing seasons give introduced plants an advantage over native plants in Interior Alaska? *Botany, 97*(6), 347–362. https://doi.org/10.1139/cjb-2018-0209

Muller, G. C., Junnila, A., Traore, M. M., Traore, S. F., Doumbia, S., Sissoko, F., Dembele, S. M., Schlein, Y., Arheart, K. L., Revay, E. E., Kravchenko, V. D., Witt, A., & Beier, J. C. (2017). The invasive shrub *Prosopis juliflora* enhances the malaria parasite transmission capacity of *Anopheles* mosquitoes: A habitat manipulation experiment. *Malaria Journal, 16*(1), 237. https://doi.org/10.1186/s12936-017-1878-9

Muñoz-Mas, R., & García-Berthou, E. (2020). Alien animal introductions in Iberian inland waters: An update and analysis. *Science of the Total Environment, 703*, 134505. https://doi.org/10.1016/j.scitotenv.2019.134505

Munyaradzi, C., & Mohamed-Katerere, J. (2006). *Africa Environment Outlook 2: Our Environment, Our Wealth. Section 3—Emerging challenges, Chapter 10: Invasive Alien Species.* (pp. 331–349). United Nations Environment Programme. https://wedocs.unep.org/20.500.11822/9626

Murchie, A. K., & Gordon, A. W. (2013). The impact of the 'New Zealand flatworm', *Arthurdendyus triangulatus*, on earthworm populations in the field. *Biological Invasions, 15*(3), 569–586. https://doi.org/10.1007/s10530-012-0309-7

Murray, B. R., & Phillips, M. L. (2012). Temporal introduction patterns of invasive alien plant species to Australia. *NeoBiota, 13*, 1–14. https://doi.org/10.3897/neobiota.13.2422

Mwangi, E., & Swallow, B. (2008). *Prosopis juliflora* invasion and rural livelihoods in the Lake Baringo area of Kenya. *Conservation and Society, 6*(2), 130–140. https://doi.org/10.4103/0972-4923.49207

Nahrung, H. F., & Carnegie, A. J. (2020). Non-native Forest Insects and Pathogens in Australia: Establishment, Spread, and Impact. *Frontiers in Forests and Global Change, 3*, 37. https://doi.org/10.3389/ffgc.2020.00037

Naughton-Treves, L., Holland, M. B., & Brandon, K. (2005). The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources, 30*(1), 219–252. https://doi.org/10.1146/annurev.energy.30.050504.164507

Naylor, R. L., Williams, S. L., & Strong, D. R. (2001). Aquaculture—A Gateway for Exotic Species. *Science, 294*(5547), 1655–1656. https://doi.org/10.1126/science.1064875

Ndhlovu, T., Milton-Dean, S. J., & Esler, K. J. (2011). Impact of *Prosopis* (mesquite) invasion and clearing on the grazing capacity of semiarid Nama Karoo rangeland, South Africa. *African Journal of Range & Forage Science, 28*(3), 129–137. https://doi.org/10.2989/10220119.2011.642095

Nealis, V. G., DeMerchant, I., Langor, D., Noseworthy, M. K., Pohl, G., Porter, K., Shanks, E., Turnquist, R., & Waring, V. (2016). Historical occurrence of alien arthropods and pathogens on trees in Canada. *Canada Journal of Forest Research, 46*(2), 172–180. https://doi.org/10.1139/cjfr-2015-0273

Neill, P. E., Rozbaczylo, N., Villaseñor-Parada, C., Guzmán-Rendón, G., Sampértegui, S., & Hernández, C. E. (2020). Patterns of association of native and exotic boring polychaetes on the southeastern Pacific coast of Chile: The combined importance of negative, positive and random interactions. *PeerJ, 8*, e8560. https://doi.org/10.7717/peerj.8560

Nelson, W. A., Sutherland, J. E., Hwang, M. S., & Choi, H.-G. (2014). New distributional record for *Pyropia koreana*: Confirmed to occur on the South Island, New Zealand. *Algae, 29*(3), 177–181. https://doi.org/10.4490/algae.2014.29.3.177

Nentwig, W. (2015). Introduction, establishment rate, pathways and impact of spiders alien to Europe. *Biological Invasions, 17*(9), 2757–2778. https://doi.org/10.1007/s10530-015-0912-5

Newman, J., Poirot, C., Roper-Gee, R., Leihy, R. I., & Chown, S. L. (2018). A decade of invertebrate colonization pressure on Scott Base in the Ross Sea region. *Biological Invasions, 20*(9), 2623–2633. https://doi.org/10.1007/s10530-018-1722-3

Nhoybouakong, M., & Khamphouke, K. (2003). Laos. In N. Pallewatta, J. K. Reaser, & A. T. Gutierrez (Eds.), *Invasive Alien Species in South-Southeast Asia: National Reports & Directory of Resources* (pp. 33–42). Global Invasive Species Programme. https://www.doi.gov/sites/doi.gov/files/uploads/invasive_alien_species_in_south_southeast_asia.pdf

Nielsen, U. N., & Wall, D. H. (2013). The future of soil invertebrate communities in polar regions: Different climate change responses in the Arctic and Antarctic? *Ecology Letters, 16*(3), 409–419. https://doi.org/10.1111/ele.12058

Nimmo, D. G., & Miller, K. K. (2007). Ecological and human dimensions of management of feral horses in Australia: A review. *Wildlife Research, 34*(5), 408–417. https://doi.org/10.1071/WR06102

Nishida, G. M. (Ed.). (2002). *Hawaiian terrestrial arthropod checklist* (4th edition). Bishop Museum.

Njiru, J., van der Knaap, M., Kundu, R., & Nyamweya, C. (2018). Lake Victoria fisheries: Outlook and management. *Lakes & Reservoirs: Research & Management, 23*(2), 152–162. https://doi.org/10.1111/lre.12220

Nnadi, N. E., & Carter, D. A. (2021). Climate change and the emergence of fungal pathogens. *PLoS Pathogens, 17*(4), e1009503. https://doi.org/10.1371/journal.ppat.1009503

NOAA. (2021). *GLANSIS (Great Lakes Aquatic Nonindigenous Species Information System).* National Oceanic and Atmospheric Administration. https://www.glerl.noaa.gov/glansis/

Noble, A., Palmer, S. M., Glaves, D. J., Crowle, A., & Holden, J. (2017). Impacts of peat bulk density, ash deposition and rainwater chemistry on establishment of peatland mosses. *Plant and Soil, 419*(1–2), 41–52. https://doi.org/10.1007/s11104-017-3325-7

Normand, S., Randin, C., Ohlemüller, R., Bay, C., Høye, T. T., Kjær, E. D., Körner, C., Lischke, H., Maiorano, L., Paulsen, J., Pearman, P. B., Psomas, A., Treier, U. A., Zimmermann, N. E., & Svenning, J.-C. (2013). A greener Greenland? Climatic potential and long-term constraints on future expansions of trees and shrubs. *Philosophical Transactions of the Royal Society B-Biological Sciences, 368*(1624). https://doi.org/10.1098/rstb.2012.0479

Novoa, A., Le Roux, J. J., Richardson, D. M., & Wilson, J. R. U. (2017). Level of environmental threat posed by horticultural trade in Cactaceae. *Conservation Biology, 31*(5), 1066–1075. https://doi.org/10.1111/cobi.12892

Novoa, A., Le Roux, J. J., Robertson, M. P., Wilson, J. R. U., & Richardson, D. M. (2015). Introduced and invasive cactus species: A global review. *AoB PLANTS, 7*, plu078. https://doi.org/10.1093/aobpla/plu078

Nowak, D. J., Pasek, J. E., Sequeira, R. A., Crane, D. E., & Mastro, V. C. (2001). Potential Effect of *Anoplophora glabripennis* (Coleoptera: Cerambycidae) on Urban Trees in the United States. *Journal of Economic Entomology, 94*(1), 116–122. https://doi.org/10.1603/0022-0493-94.1.116

Nunes, A. L., Tricarico, E., Panov, V. E., Cardoso, A. C., & Katsanevakis, S. (2015). Pathways and gateways of freshwater invasions in Europe. *Aquatic Invasions, 10*(4), 359–370. https://doi.org/10.3391/ai.2015.10.4.01

Nuñez, M. A., & Amano, T. (2021). Monolingual searches can limit and bias results in global literature reviews. *Nature Ecology & Evolution, 5*(3), 264. https://doi.org/10.1038/s41559-020-01369-w

Nuñez, M. A., Chiuffo, M. C., Seebens, H., Kuebbing, S., McCary, M. A., Lieurance, D., Zhang, B., Simberloff, D., & Meyerson, L. A. (2021). Two decades of data reveal that Biological Invasions needs to increase participation beyond North America, Europe, and Australasia. *Biological Invasions, 24*, 333–340. https://doi.org/10.1007/s10530-021-02666-6

Nunez-Mir, G. C., Liebhold, A. M., Guo, Q., Brockerhoff, E. G., Jo, I., Ordonez, K., & Fei, S. (2017). Biotic resistance to exotic invasions: Its role in forest ecosystems, confounding artifacts, and future directions. *Biological Invasions, 19*(11), 3287–3299. https://doi.org/10.1007/s10530-017-1413-5

Nyári, Á., Ryall, C., & Peterson, A. T. (2006). Global invasive potential of the house crow *Corvus splendens* based on ecological niche modelling. *Journal of Avian Biology, 37*(4), 306–311. https://doi.org/10.1111/j.2006.0908-8857.03686.x

Obua, J., Agea, J. G., & Ogwal, J. J. (2010). Status of forests in Uganda. *African Journal of Ecology, 48*(4), 853–859. https://doi.org/10.1111/j.1365-2028.2010.01217.x

O'Dowd, D. J., Green, P. T., & Lake, P. S. (2003). Invasional 'meltdown' on an oceanic island. *Ecology Letters, 6*(9), 812–817. https://doi.org/10.1046/j.1461-0248.2003.00512.x

Oduor, A. M. O. (2013). Evolutionary responses of native plant species to invasive plants: A review. *New Phytologist, 200*(4), 986–992. https://doi.org/10.1111/nph.12429

O'Flynn, C., Kelly, J., & Lysaght, L. (2014). *Ireland's invasive and non-native species – trends in introductions* (2; National Biodiversity Data Centre Series, p. 50). The National Biodiversity Data Centre. https://biodiversityireland.ie/app/uploads/2021/06/Trends-Report-2013.pdf

O'Hanlon, S. J., Rieux, A., Farrer, R. A., Rosa, G. M., Waldman, B., Bataille, A., Kosch, T. A., Murray, K. A., Brankovics, B., Fumagalli, M., Martin, M. D., Wales, N., Alvarado-Rybak, M., Bates, K. A., Berger, L., Böll, S., Brookes, L., Clare, F., Courtois, E. A., … Fisher, M. C. (2018). Recent Asian origin of chytrid fungi causing global amphibian declines. *Science, 360*(6389), 621–627. https://doi.org/10.1126/science.aar1965

Ojaveer, H., Galil, B. S., Campbell, M. L., Carlton, J. T., Canning-Clode, J., Cook, E. J., Davidson, A. D., Hewitt, C. L., Jelmert, A., Marchini, A., McKenzie, C. H., Minchin, D., Occhipinti-Ambrogi, A., Olenin, S., & Ruiz, G. (2015). Classification of non-indigenous species based on their impacts: Considerations for application in marine management. *PLoS Biology, 13*(4), e1002130. https://doi.org/10.1371/journal.pbio.1002130

Ojaveer, H., Galil, B. S., Carlton, J. T., Alleway, H., Goulletquer, P., Lehtiniemi, M., Marchini, A., Miller, W., Occhipinti-Ambrogi, A., Peharda, M., Ruiz, G. M., Williams, S. L., & Zaiko, A. (2018). Historical baselines in marine bioinvasions: Implications for policy and management. *PLoS ONE, 13*(8), e0202383. https://doi.org/10.1371/journal.pone.0202383

Ojaveer, H., Galil, B. S., Gollasch, S., Marchini, A., Minchin, D., Occhipinti-Ambrogi, A., & Olenin, S. (2014). Identifying the top issues of marine invasive alien species in Europe. *Management of Biological Invasions, 5*(2), 81–84. https://doi.org/10.3391/mbi.2014.5.2.01

Ojaveer, H., Olenin, S., Narščius, A., Florin, A. B., Ezhova, E., Gollasch, S., Jensen, K. R., Lehtiniemi, M., Minchin, D., Normant-Saremba, M., & Strāke, S. (2017). Dynamics of biological invasions and pathways over time: A case study of a temperate coastal sea. *Biological Invasions, 19*(3), 799–813. https://doi.org/10.1007/s10530-016-1316-x

Olenin, S., Didžiulis, V., Ovčarenko, I., Olenina, I., Nunn, A. D., & Cowx, I. G. (2008). *Review of introductions of aquatic species in Europe* (EC FP6 Coordination Action IMPASSE, Technical Report 044142; p. 65). Coastal Research and Planning Institute, Klaipėda University (KUCORPI). https://doi.org/10.13140/RG.2.1.4504.7449

Orlov, Yu. I., & Ivanov, B. G. (1978). On the introduction of the Kamchatka King crab *Paralithodes camtschatica* (Decapoda: Anomura: Lithodidae) into the Barents Sea. *Marine Biology, 48*(4), 373–375. https://doi.org/10.1007/BF00391642

Orlova, M. I., Therriault, T. W., Antonov, P. I., & Shcherbina, G. Kh. (2005). Invasion ecology of quagga mussels (*Dreissena rostriformis bugensis*): A review of evolutionary and phylogenetic impacts. *Aquatic Ecology, 39*(4), 401–418. https://doi.org/10.1007/s10452-005-9010-6

Oróstica, M. H., Otaíza, R. D., & Neill, P. E. (2012). Blades and papillae as likely dispersing propagules in Chilean populations of *Mastocarpus* sp. (Rhodophyta, Gigartinales). *Revista de Biología Marina y Oceanografía, 47*(No1), 109–119. http://repositoriodigital.ucsc.cl/bitstream/handle/25022009/550/Mauricio%20H.%20Or%C3%B3stica%2C%20Ricardo%20D.%20Ota%C3%ADza%2C%20Paula%20E.%20Neill.pdf?sequence=1&isAllowed=y

Ortíz-Ceballos, A. I., Ortiz-Gamino, D., Andrade-Torres, A., Pérez-Rodríguez, P., & López-Ortega, M. (2019). *Pontoscolex corethrurus*: A homeless invasive tropical earthworm? *PLoS ONE, 14*(9), e0222337. https://doi.org/10.1371/journal.pone.0222337

Osborne, M. A. (2000). Acclimatizing the World: A History of the Paradigmatic Colonial Science. *Osiris, 15*(1), 135–151. https://doi.org/10.1086/649323

Osipova, E., Shadie, P., Zwahlen, C., Osti, M., Shi, Y., Kormos, C., Bertzky, B., Van Murai, M., Merm, R., & Badman, T. (2017). *IUCN World Heritage Outlook 2: A conservation assessment of all natural World Heritage sites* (p. 92). International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2017.17.en

Osyczka, P. (2010). Alien lichens unintentionally transported to the "Arctowski" station (South Shetlands, Antarctica). *Polar Biology, 33*(8), 1067–1073. https://doi.org/10.1007/s00300-010-0786-z

Osyczka, P., Mleczko, P., Karasiński, D., & Chlebicki, A. (2012). Timber transported to Antarctica: A potential and undesirable carrier for alien fungi and insects. *Biological Invasions, 14*(1), 15–20. https://doi.org/10.1007/s10530-011-9991-0

Otfinowski, R., Kenkel, N. C., & Catling, P. M. (2007). The biology of Canadian weeds. 134. *Bromus inermis* Leyss. *Canadian Journal of Plant Science, 87*(1), 183–198. https://doi.org/10.4141/P06-071

Oualid, J. A., Iazza, B., Tamsouri, N. M., El Aamri, F., Moukrim, A., & López–González, P. J. (2019). Hidden diversity under morphology–based identifications of widespread invasive species: The case of the "well–known" hydromedusa *Craspedacusta sowerbii* Lankester 1880. *Animal Biodiversity and Conservation, 42*(2), 301–316. https://doi.org/10.32800/abc.2019.42.0301

Paap, T., Burgess, T. I., & Wingfield, M. J. (2017). Urban trees: Bridge-heads for forest pest invasions and sentinels for early detection. *Biological Invasions, 19*(12), 3515–3526. https://doi.org/10.1007/s10530-017-1595-x

Packer, J. G., Meyerson, L. A., Richardson, D. M., Brundu, G., Allen, W. J., Bhattarai, G. P., Brix, H., Canavan, S., Castiglione, S., Cicatelli, A., Čuda, J., Cronin, J. T., Eller, F., Guarino, F., Guo, W.-H., Guo, W.-Y., Guo, X., Hierro, J. L., Lambertini, C., … Pyšek, P. (2017). Global networks for invasion science: Benefits, challenges and guidelines. *Biological Invasions, 19*(4), 1081–1096. https://doi.org/10.1007/s10530-016-1302-3

Paclibar, G. C. B., & Tadiosa, E. R. (2019). Ecological niche modeling of invasive alien plant species in a protected landscape. *Global Journal of Environmental Science and Management, 5*(3), 371–382. https://doi.org/10.22034/gjesm.2019.03.09

Padayachee, A. L., Irlich, U. M., Faulkner, K. T., Gaertner, M., Procheş, Ş., Wilson, J. R. U., & Rouget, M. (2017). How do invasive species travel to and through urban environments? *Biological Invasions, 19*(12), 3557–3570. https://doi.org/10.1007/s10530-017-1596-9

Padmanaba, M., Tomlinson, K. W., Hughes, A. C., & Corlett, R. T. (2017). Alien plant invasions of protected areas in Java, Indonesia. *Scientific Reports, 7*(1), 9334. https://doi.org/10.1038/s41598-017-09768-z

Pagad, S., Bisset, S., Genovesi, P., Groom, Q., Hirsch, T., Jetz, W., Ranipeta, A., Schigel, D., Sica, Y. V., & McGeoch, M. A. (2022). Country Compendium of the Global Register of Introduced and Invasive Species. *Scientific Data, 9*(1), 391. https://doi.org/10.1038/s41597-022-01514-z

Pagnucco, K. S., Maynard, G. A., Fera, S. A., Yan, N. D., Nalepa, T. F., & Ricciardi, A. (2015). The Future of species invasions in the Great Lakes-St. Lawrence River basin. *Journal of Great Lakes Research, 41*(S1), 96–107. https://doi.org/10.1016/j.jglr.2014.11.004

Paiaro, V., Cabido, M., & Pucheta, E. (2011). Altitudinal distribution of native and alien plant species in roadside communities from central Argentina. *Austral Ecology, 36*(2), 176–184. https://doi.org/10.1111/j.1442-9993.2010.02134.x

Palacio-López, K., & Gianoli, E. (2011). Invasive plants do not display greater phenotypic plasticity than their native or non-invasive counterparts: A meta-analysis. *Oikos, 120*(9), 1393–1401. https://doi.org/10.1111/j.1600-0706.2010.19114.x

Pallewatta, N., Reaser, J. K., & Gutierrez, A. T. (Eds.). (2003). Sri Lanka. In *Invasive Alien Species in South-Southeast Asia: National Reports & Directory of Resources* (p. 111). Global Invasive Species Programme. https://www.doi.gov/sites/doi.gov/files/uploads/invasive_alien_species_in_south_southeast_asia.pdf

Palma, E., Catford, J. A., Corlett, R. T., Duncan, R. P., Hahs, A. K., McCarthy, M. A., McDonnell, M. J., Thompson, K., Williams, N. S. G., & Vesk, P. A. (2017). Functional trait changes in the floras of 11 cities across the globe in response to urbanization. *Ecography, 40*(7), 875–886. https://doi.org/10.1111/ecog.02516

Pan, X., Villamagna, A. M., & Li, B. (2011). *Eichhornia crassipes* Mart. (Solms-Laubach) (water hyacinth). In R. A. Francis (Ed.), *A Handbook of Global Freshwater Invasive Species.* Routledge. https://doi.org/10.4324/9780203127230

Parent, G., Thoen, D., & Calonge, F. D. (2000). Nouvelles données sur la répartition de *Clathrus archeri*, en particulier dans l'ouest et le sud-ouest de l' Europe. *Bulletin Trimestriel de La Société Mycologique de France, 116*(3), 241–266. https://www.semanticscholar.org/paper/Nouvelles-donn%C3%A9es-sur-la-r%C3%A9partition-de-Clathrus-en-Parent-Thoen/6fce268b1aea10772c7ec18fcc2b3aaa1153d146

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeva, T., Silvius, M., & Stringer, L. (2008). *Assessment on peatlands, biodiversity and climate change: Main report.* (p. 179). Global Environment Centre, Kuala Lumpur & Wetlands International, Wageningen. https://globalclimateactionpartnership.org/app/uploads/2015/12/Assessment-on-peatlands-biodiversity-and-climate-change-main-report.pdf

Parker, J. D., Torchin, M. E., Hufbauer, R. A., Lemoine, N. P., Alba, C., Blumenthal, D. M., Bossdorf, O., Byers, J. E., Dunn, A. M., Heckman, R. W., Hejda, M., Jarošík, V., Kanarek, A. R., Martin, L. B., Perkins, S. E., Pyšek, P., Schierenbeck, K., Schlöder, C., van Klinken, R., … Wolfe, L. M. (2013). Do invasive species perform better in their new ranges? *Ecology, 94*(5), 985–994. https://doi.org/10.1890/12-1810.1

Parkes, J. P., Byrom, A. E., Landcare Research, & Edge, K.-A. (2017). Eradicating mammals on New Zealand island reserves: What is left to do? *New Zealand Journal of Ecology, 41*(2), 263–270. https://doi.org/10.20417/nzjecol.41.25

Parkinson, A. J., & Butler, J. C. (2005). Potential impacts of climate change on infectious diseases in the Arctic. *International Journal of Circumpolar Health, 64*(5), 478–486. https://doi.org/10.3402/ijch.v64i5.18029

Pascal, M. (1980). Structure et dynamique de la population de chats harets de l'archipel des Kerguelen. *Mammalia, 44*(2), 161–182. https://doi.org/10.1515/mamm.1980.44.2.161

Patnaik, P., Abbasi, T., & Abbasi, S. A. (2017). Prosopis *(Prosopis juliflora)*: Blessing and bane. *Tropical Ecology, 58*(3), 455–483.

Pattison, Z., Minderman, J., Boon, P. J., & Willby, N. (2017). Twenty years of change in riverside vegetation: What role have invasive alien plants played? *Applied Vegetation Science, 20*(3), 422–434. https://doi.org/10.1111/avsc.12297

Patzelt, A., Pyšek, P., Pergl, J., & van Kleunen, M. (2022). Alien flora of Oman: Invasion status, taxonomic composition, habitats, origin, and pathways of introduction. *Biological Invasions, 24*(4), 955–970. https://doi.org/10.1007/s10530-021-02711-4

Pauchard, A., Aguayo, M., Peña, E., & Urrutia, R. (2006). Multiple effects of urbanization on the biodiversity of developing countries: The case of a fast-growing metropolitan area (Concepción, Chile). *Biological Conservation, 127*(3), 272–281. https://doi.org/10.1016/j.biocon.2005.05.015

Pauchard, A., & Alaback, P. B. (2004). Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile. *Conservation Biology, 18*(1), 238–248. https://doi.org/10.1111/j.1523-1739.2004.00300.x

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J., Edwards, P. J., Arévalo, J. R., Cavieres, L. A., Guisan, A., Haider, S., Jakobs, G., McDougall, K., Millar, C. I., Naylor, B. J., Parks, C. G., Rew, L. J., & Seipel, T. (2009). Ain't no mountain high enough: Plant invasions reaching new elevations. *Frontiers in Ecology and the Environment, 7*(9), 479–486. https://doi.org/10.1890/080072

Pauchard, A., Milbau, A., Albihn, A., Alexander, J., Burgess, T., Daehler, C., Englund, G., Essl, F., Evengård, B., Greenwood, G. B., Haider, S., Lenoir, J., McDougall, K., Muths, E., Nuñez, M. A., Olofsson, J., Pellissier, L., Rabitsch, W., Rew, L. J., … Kueffer, C. (2016). Non-native and native organisms moving into high elevation and high latitude ecosystems in an era of climate change: New challenges for ecology and conservation. *Biological Invasions, 18*(2), 345–353. https://doi.org/10.1007/s10530-015-1025-x

Paulay, G., Kirkendale, L., Lambert, G., & Meyer, C. (2002). Anthropogenic biotic interchange in a coral reef ecosystem: A case study from Guam. *Pacific Science, 56*(4), 403–422. https://doi.org/10.1353/psc.2002.0036

Pavlov, N. V. (Ed.). (1956). *Flora Kazakhstana*. Izd-vo Akademii nauk Kazakhskoĭ SSR. http://books.google.com/books?id=EeklAQAAMAAJ

Peacock, L., & Worner, S. (2006). Using analogous climates and global insect distribution data to identify potential sources of new invasive insect pests in New Zealand. *New Zealand Journal of Zoology, 33*(2), 141–145. https://doi.org/10.1080/03014223.2006.9518438

Pearman, P. B., Guisan, A., Broennimann, O., & Randin, C. F. (2008). Niche dynamics in space and time. *Trends in Ecology & Evolution, 23*(3), 149–158. https://doi.org/10.1016/j.tree.2007.11.005

Peck, S. B., Heraty, J., Landry, B., & Sinclair, B. J. (1998). Introduced insect fauna of an oceanic archipelago: The Galápagos Islands, Ecuador. *American Entomologist, 44*(4), 218–237. https://doi.org/10.1093/ae/44.4.218

Pedrosa, M. C., Borges, M. A. Z., Eiras, Á. E., Caldas, S., Cecílio, A. B., Brito, M. F., & Ribeiro, S. P. (2020). Invasion of tropical montane cities by *Aedes aegypti* and *Aedes albopictus* (Diptera: Culicidae) depends on continuous warm winters and suitable urban biotopes. *Journal of Medical Entomology, 58*(1), 333–342. https://doi.org/10.1093/jme/tjaa135

Peeler, E. J., Oidtmann, B. C., Midtlyng, P. J., Miossec, L., & Gozlan, R. E. (2011). Non-native aquatic animals introductions have driven disease emergence in Europe. *Biological Invasions, 13*(6), 1291–1303. https://doi.org/10.1007/s10530-010-9890-9

Peh, K. S.-H. (2010). Invasive species in Southeast Asia: The knowledge so far. *Biodiversity and Conservation, 19*(4), 1083–1099. https://doi.org/10.1007/s10531-009-9755-7

Pellerin, S., & Lavoie, C. (2000). Peatland fragments of southern Quebec: Recent evolution of their vegetation structure. *Canadian Journal of Botany, 78*(2), 255–265. https://doi.org/10.1139/b99-186

Pelser, P. B., Barcelona, J. F., & Nickrent, D. L. (2011). *Co's Digital Flora of the Philippines*. http://www.philippineplants.org

Peltanová, A., Petrusek, A., Kment, P., & Juřičková, L. (2012). A fast snail's pace: Colonization of Central Europe by Mediterranean gastropods. *Biological Invasions, 14*(4), 759–764. https://doi.org/10.1007/s10530-011-0121-9

Peña, E., Hidalgo, M., Langdon, B., & Pauchard, A. (2008). Patterns of spread of *Pinus contorta* Dougl. Ex Loud. Invasion in a Natural Reserve in southern South America. *Forest Ecology and Management, 256*(5), 1049–1054. https://doi.org/10.1016/j.foreco.2008.06.020

Perella, C. D., & Behm, J. E. (2020). Understanding the spread and impact of exotic geckos in the greater Caribbean region. *Biodiversity and Conservation, 29*(4), 1109–1134. https://doi.org/10.1007/s10531-020-01939-1

Pérez, J. E., Alfonsi, C., Salazar, S. K., Macsotay, O., Barrios, J., & Martínez Escarbassiere, R. (2007). Especies marinas exóticas y criptogénicas en las costas de Venezuela. *Boletín Del Instituto Oceanográfico de Venezuela, 46*(1), 79–96.

Pérez-Postigo, I., Bendix, J., Vibrans, H., & Cuevas-Guzmán, R. (2021). Diversity of alien roadside herbs along an elevational gradient in western Mexico. *NeoBiota, 65*, 71–91. https://doi.org/10.3897/neobiota.65.67192

Pertierra, L. R., Hughes, K. A., Tejedo, P., Enriquez, N., Lucianez, M. J., & Benayas, J. (2017). Eradication of the non-native *Poa pratensis* colony at Cierva Point, Antarctica: A case study of international cooperation and practical management in an area under multi-party governance. *Environmental Science & Policy, 69*, 50–56. https://doi.org/10.1016/j.envsci.2016.12.009

Perzanowska, J., Korzeniak, J., & Chmura, D. (2019). Alien species as a potential threat for Natura 2000 habitats: A national survey. *PeerJ, 7*, e8032. https://doi.org/10.7717/peerj.8032

Pestana, L. B., Dias, G. M., & Marques, A. C. (2017). A century of introductions by coastal sessile marine invertebrates in Angola, South East Atlantic Ocean. *Marine Pollution Bulletin, 125*(1–2), 426–432. https://doi.org/10.1016/j.marpolbul.2017.09.041

Petitpierre, B., McDougall, K., Seipel, T., Broennimann, O., Guisan, A., & Kueffer, C. (2016). Will climate change increase the risk of plant invasions into mountains? *Ecological Applications, 26*(2), 530–544. https://doi.org/10.1890/14-1871

Petraitis, P. S. (1987). Factors organizing rocky intertidal communities of New England: Herbivory and predation in sheltered bays. *Journal of Experimental Marine Biology and Ecology, 109*(2), 117–136. https://doi.org/10.1016/0022-0981(87)90011-6

Phillips, B. L., Brown, G. P., Webb, J. K., & Shine, R. (2006). Invasion and the evolution of speed in toads. *Nature, 439*(16), 803. https://doi.org/10.1038/439803a

Pichlová-Ptáčníková, R., & Vanderploeg, H. A. (2009). The invasive cladoceran *Cercopagis pengoi* is a generalist predator capable of feeding on a variety of prey species of different sizes and escape abilities. *Fundamental and Applied Limnology/Archiv Für Hydrobiologie, 173*(4), 267–279. https://doi.org/10.1127/1863-9135/2009/0173-0267

Picker, M. D., & Griffiths, C. L. (2017). Alien animals in South Africa – composition, introduction history, origins and distribution patterns. *Bothalia, 47*(2), a2147. https://doi.org/10.4102/abc.v47i2.2147

Pickering, C. M., Bear, R., & Hill, W. (2007). Indirect impacts of nature based tourism and recreation: The association between infrastructure and the diversity of exotic plants in Kosciuszko National Park, Australia. *Journal of Ecotourism, 6*(2), 146–157. https://doi.org/10.2167/joe162.0

Pili, A. N., Tingley, R., Sy, E. Y., Diesmos, M. L. L., & Diesmos, A. C. (2020). Niche shifts and environmental non-equilibrium undermine the usefulness of ecological niche models for invasion risk assessments. *Scientific Reports, 10*(1), 7972. https://doi.org/10.1038/s41598-020-64568-2

Pipek, P., Pyšek, P., & Blackburn, T. M. (2015). How the Yellowhammer became a Kiwi: The history of an alien bird invasion revealed. *NeoBiota, 24*, 1–31. https://doi.org/10.3897/neobiota.24.8611

Pitcher, T. J., & Hart, P. J. B. (Eds.). (1995). *The Impact of Species Changes in African Lakes* (Vol. 18). Springer Netherlands. https://doi.org/10.1007/978-94-011-0563-7

Pivello, V. R., Shida, C. N., & Meirelles, S. T. (1999). Alien grasses in Brazilian savannas: A threat to the biodiversity. *Biodiversity & Conservation, 8*(9), 1281–1294. https://doi.org/10.1023/A:1008933305857

Pleguezuelos, J. M. (2002). Las especies introducidas de anfibios y reptiles. In J. M. Pleguezuelos, R. Márquez, & M. Lizana (Eds.), *Atlas y libro rojo de los anfibios y reptiles de España* (pp. 501–532). Dirección General de Conservación de la Naturaleza-Asociación Herpetologica Española.

Pociecha, A., Solarz, W., Najberek, K., & Wilk-Woźniak, E. (2016). Native, alien, cosmopolitan, or cryptogenic? A framework for clarifying the origin status of rotifers. *Aquatic Biology, 24*(3), 141–149. https://doi.org/10.3354/ab00644

Poessel, S. A., Beard, K. H., Callahan, C. M., Ferreira, R. B., & Stevenson, E. T. (2012). Biotic acceptance in introduced amphibians and reptiles in Europe and North America. *Global Ecology and Biogeography, 22*(2), 192–201. https://doi.org/10.1111/j.1466-8238.2012.00797.x

Poland, T. M., & McCullough, D. G. (2006). Emerald Ash Borer: Invasion of the Urban Forest and the Threat to North America's Ash Resource. *Journal of Forestry, 104*(3), 118–124. https://academic.oup.com/jof/article/104/3/118/4598702

Ponge, J. (2021). Communities, ecosystem engineers, and functional domains. *Ecological Research, 36*(5), 766–777. https://doi.org/10.1111/1440-1703.12247

Porco, D., Decaëns, T., Deharveng, L., James, S. W., Skarżyński, D., Erséus, C., Butt, K. R., Richard, B., & Hebert, P. D. N. (2013). Biological invasions in soil: DNA barcoding as a monitoring tool in a multiple taxa survey targeting European earthworms and springtails in North America. *Biological Invasions, 15*(4), 899–910. https://doi.org/10.1007/s10530-012-0338-2

Porter, S. S., Faber-Hammond, J. J., & Friesen, M. L. (2018). Co-invading symbiotic mutualists of *Medicago polymorpha* retain high ancestral diversity and contain diverse accessory genomes. *FEMS Microbiology Ecology, 94*(1), fix168. https://doi.org/10.1093/femsec/fix168

Potapov, A., Schaefer, I., Jochum, M., Widyastuti, R., Eisenhauer, N., & Scheu, S. (2021). Oil palm and rubber expansion facilitates earthworm invasion in Indonesia. *Biological Invasions, 23*(9), 2783–2795. https://doi.org/10.1007/s10530-021-02539-y

Poursanidis, D., Kalogirou, S., Azzurro, E., Parravicini, V., Bariche, M., & zu Dohna, H. (2020). Habitat suitability, niche unfilling and the potential spread of *Pterois miles* in the Mediterranean Sea. *Marine Pollution Bulletin, 154*, 111054. https://doi.org/10.1016/j.marpolbul.2020.111054

Pouteau, R., & Birnbaum, P. (2016). Island biodiversity hotspots are getting hotter: Vulnerability of tree species to climate change in New Caledonia. *Biological Conservation, 201*, 111–119. https://doi.org/10.1016/j.biocon.2016.06.031

Powell, R., Henderson, R. W., Farmer, M. C., Breuil, M., Echternacht, A. C., van Buurt, G., Romagosa, C. M., & Perry, G. (2011). Introduced amphibians and reptiles in the Greater Caribbean: Patterns and conservation implications. In A. Hailey, B. S. Wilson, & J. A. Horrocks (Eds.), *Conservation of Caribbean Island Herpetofaunas, Conservation Biology and the Wider Caribbean* (Vol. 1, pp. 63–143). Brill. https://doi.org/10.1163/ej.9789004183957.i-228.38

Preston, C. D., Pearman, D. A., & Dines, T. D. (2002). *New atlas of the British and Irish flora.* (p. 922). Oxford University Press.

Preston, C. D., Pearman, D. A., & Hall, A. R. (2004). Archaeophytes in Britain. *Botanical Journal of the Linnean Society, 145*(3), 257–294. https://doi.org/10.1111/j.1095-8339.2004.00284.x

Pretorius, Z. A., Bender, C. M., Visser, B., & Terefe, T. (2010). First Report of a *Puccinia graminis* f. Sp. *Tritici* Race Virulent to the *Sr24* and *Sr31* Wheat Stem Rust Resistance Genes in South Africa. *Plant Disease, 94*(6), 784–784. https://doi.org/10.1094/PDIS-94-6-0784C

Priddel, D., Carlile, N., & Wheeler, R. (2000). Eradication of European rabbits *(Oryctolagus cuniculus)* from Cabbage Tree Island, NSW, Australia, to protect the breeding habitat of Gould's petrel *(Pterodroma leucoptera leucoptera). Biological Conservation, 94*(1), 115–125. https://doi.org/10.1016/S0006-3207(99)00155-X

Priede, I. G., & Froese, R. (2013). Colonization of the deep sea by fishes. *Journal of Fish Biology, 83*(6), 1528–1550. https://doi.org/10.1111/jfb.12265

Pringle, A., Bever, J. D., Gardes, M., Parrent, J. L., Rillig, M. C., & Klironomos, J. N. (2009). Mycorrhizal Symbioses and Plant Invasions. *Annual Review of Ecology, Evolution, and Systematics, 40*(1), 699–715. https://doi.org/10.1146/annurev.ecolsys.39.110707.173454

Pringle, A., & Vellinga, E. C. (2006). Last chance to know? Using Literature to explore the biogeography and invasion biology of the death cap mushroom *Amanita phalloides* (Vaill. ex Fr. :Fr.) Link. *Biological Invasions, 8*(5), 1131–1144. https://doi.org/10.1007/s10530-005-3804-2

Pringle, R. M. (2005). The Origins of the Nile Perch in Lake Victoria. *BioScience, 55*(9), 780–787. https://doi.org/10.1641/0006-3568(2005)055[0780:TOOTNP]2.0.CO;2

Puckett, E. E., Orton, D., & Munshi-South, J. (2020). Commensal rats and humans: Integrating rodent phylogeography and zooarchaeology to highlight connections between human societies. *BioEssays, 42*(5), 1900160. https://doi.org/10.1002/bies.201900160

Pyšek, P. (1998). Alien and native species in Central European urban floras: A quantitative comparison. *Journal of Biogeography, 25*(1), 155–163. https://doi.org/10.1046/j.1365-2699.1998.251177.x

Pyšek, P., Bacher, S., Chytrý, M., Jarošík, V., Wild, J., Celesti-Grapow, L., Gassó, N., Kenis, M., Lambdon, P. W., Nentwig, W., Pergl, J., Roques, A., Sádlo, J., Solarz, W., Vilà, M., & Hulme, P. E. (2010). Contrasting patterns in the invasions of European terrestrial and freshwater habitats by alien plants, insects and vertebrates. *Global Ecology and Biogeography, 19*(3), 317–331. https://doi.org/10.1111/j.1466-8238.2009.00514.x

Pyšek, P., Blackburn, T. M., García-Berthou, E., Perglová, I., & Rabitsch, W. (2017). Displacement and Local Extinction of Native and Endemic Species. In M. Vilà & P. E. Hulme (Eds.), *Impact of Biological Invasions on Ecosystem Services* (Vol. 12, pp. 157–175). Springer International Publishing. https://doi.org/10.1007/978-3-319-45121-3_10

Pyšek, P., Dawson, W., Essl, F., Kreft, H., Pergl, J., Seebens, H., van Kleunen, M., Weigelt, P., & Winter, M. (2019). Contrasting patterns of naturalized plant richness in the Americas: Numbers are higher in the North but expected to rise sharply in the South. *Global Ecology and Biogeography, 28*(6), 779–783. https://doi.org/10.1111/geb.12891

Pyšek, P., Hulme, P. E., Simberloff, D., Bacher, S., Blackburn, T. M., Carlton, J. T., Dawson, W., Essl, F., Foxcroft, L. C., Genovesi, P., Jeschke, J. M., Kühn, I., Liebhold, A. M., Mandrak, N. E., Meyerson, L. A., Pauchard, A., Pergl, J., Roy, H. E., Seebens, H., … Richardson, D. M. (2020). Scientists' warning on invasive alien species. *Biological Reviews, 95*(6), 1511–1534. https://doi.org/10.1111/brv.12627

Pyšek, P., & Jarošík, V. (2005). Residence time determines the distribution of alien plants. In Inderjit (Ed.), *Invasive Plants: Ecological and Agricultural Aspects* (pp. 77–96). Birkhäuser Verlag. https://doi.org/10.1007/3-7643-7380-6_5

Pyšek, P., Jarosik, V., Hulme, P. E., Kühn, I., Wild, J., Arianoutsou, M., Bacher, S., Chiron, F., Didžiulis, V., Essl, F., Genovesi, P., Gherardi, F., Hejda, M., Kark, S., Lambdon, P. W., Desprez-Loustau, M.-L., Nentwig, W., Pergl, J., Poboljšaj, K., … Winter, M. (2010). Disentangling the role of environmental and human pressures on biological invasions across Europe. *Proceedings of the National Academy of Sciences, 107*(27), 12157–12162. https://doi.org/10.1073/pnas.1002314107

Pyšek, P., Jarošík, V., Hulme, P. E., Pergl, J., Hejda, M., Schaffner, U., & Vilà, M. (2012). A global assessment of invasive plant impacts on resident species, communities and ecosystems: The interaction of impact measures, invading species' traits and environment. *Global Change Biology, 18*(5), 1725–1737. https://doi.org/10.1111/j.1365-2486.2011.02636.x

Pyšek, P., Jarošík, V., & Pergl, J. (2011). Alien plants introduced by different pathways differ in invasion success: Unintentional introductions as a threat to natural areas. *PLoS ONE, 6*(9), e24890. https://doi.org/10.1371/journal.pone.0024890

Pyšek, P., Pergl, J., Dawson, W., Essl, F., Kreft, H., Weigelt, P., Winter, M., & van Kleunen, M. (2020). European plant invasions. In D. R. Clements, M. K. Upadhyaya, S. Joshi, & A. Shrestha (Eds.), *Global Plant Invasions* (pp. 151–165). Springer. https://doi.org/10.1007/978-3-030-89684-3_7

Pyšek, P., Pergl, J., Essl, F., Lenzner, B., Dawson, W., Kreft, H., Weigelt, P., Winter, M., Kartesz, J., Nishino, M., Antonova, L. A., Barcelona, J. F., Cabesaz, F. J., Cárdenas, D., Cárdenas-Toro, J., Castaño, N., Chacón, E., Chatelain, C., Dullinger, S., … Kleunen, M. van. (2017). Naturalized alien flora of the world: Species diversity, taxonomic and phylogenetic patterns, geographic distribution and global hotspots of plant invasion. *Preslia, 89*(3), 203–274. https://doi.org/10.23855/preslia.2017.203

Pyšek, P., Pergl, J., van Kleunen, M., Dawson, W., Essl, F., Kreft, H., Weigelt, P., Wilson, J. R., Winter, M., & Richardson, D. M. (2020). South Africa as a Donor of Naturalised and Invasive Plants to Other Parts of the World. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 759–785). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_26

Pyšek, P., & Prach, K. (1993). Plant invasions and the role of riparian habitats: A comparison of four species alien to central Europe. *Journal of Biogeography, 20*(4), 413–420. https://doi.org/10.2307/2845589

Pyšek, P., & Richardson, D. M. (2006). The biogeography of naturalization in alien plants. *Journal of Biogeography, 33*(12), 2040–2050. https://doi.org/10.1111/j.1365-2699.2006.01578.x

Pyšek, P., Richardson, D. M., Pergl, J., Jarošík, V., Sixtová, Z., & Weber, E. (2008). Geographical and taxonomic biases in invasion ecology. *Trends in Ecology & Evolution, 23*(5), 237–244. https://doi.org/10.1016/j.tree.2008.02.002

Pyšek, P., Sádlo, J., & Mandák, B. (2002). Catalogue of alien plants of the Czech Republic. *Preslia, 74*(2), 97–186. https://www.preslia.cz/P022CPys.pdf

Qian, H. (2008). A latitudinal gradient of beta diversity for exotic vascular plant species in North America. *Diversity and Distributions, 14*(3), 556–560. https://doi.org/10.1111/j.1472-4642.2008.00466.x

Qian, H., & Ricklefs, R. E. (2000). Large-scale processes and the Asian bias in species diversity of temperate plants. *Nature, 407*(6801), 180–182. https://doi.org/10.1038/35025052

Qin, J., Cheng, F., Zhang, L., Schmidt, B. V., Liu, J., & Xie, S. (2019). Invasions of two estuarine gobiid species interactively induced from water diversion and saltwater intrusion. *Management of Biological Invasions, 10*(1), 139–150. https://doi.org/10.3391/mbi.2019.10.1.09

Qiu, J. (2013). China's cordgrass plan is 'overkill.' *Nature, 499*(7459), 392–393. https://doi.org/10.1038/499392a

Qureshi, H., Arshad, M., & Bibi, Y. (2014). Invasive flora of Pakistan: A critical analysis. *International Journal of Biosciences, 4*(1), 407–424. https://doi.org/10.12692/ijb/4.1.407-424

Rabitsch, W., & Nehring, S. (Eds.). (2017). *Naturschutzfachliche Invasivitätsbewertungen für in Deutschland wild lebende gebietsfremde aquatische Pilze, Niedere Pflanzen und Wirbellose Tiere.* Bundesamt für Naturschutz. https://doi.org/10.19217/skr458

Rago, A., While, G. M., & Uller, T. (2012). Introduction pathway and climate trump ecology and life history as predictors of establishment success in alien frogs and toads. *Ecology and Evolution, 2*(7), 1437–1445. https://doi.org/10.1002/ece3.261

Rajagopal, S., Venugopalan, V. P., van der Velde, G., & Jenner, H. A. (2006). Greening of the coasts: A review of the *Perna viridis* success story. *Aquatic Ecology, 40*(3), 273–297. https://doi.org/10.1007/s10452-006-9032-8

Ramos, R. S., Kumar, L., Shabani, F., & Picanco, M. C. (2018). Mapping global risk levels of *Bemisia tabaci* in areas of suitability for open field tomato cultivation under current and future climates. *PLoS ONE, 13*(6), e0198925. https://doi.org/10.1371/journal.pone.0198925

Ramprasad, V., Joglekar, A., & Fleischman, F. (2020). Plantations and pastoralists: Afforestation activities make pastoralists in the Indian Himalaya vulnerable. *Ecology and Society, 25*(4), 1. https://doi.org/10.5751/ES-11810-250401

Randall, R. P. (2002). *A global compendium of weeds.* R.G. and F.J. Richardson.

Rashid, I., Haq, S. M., Lembrechts, J. J., Khuroo, A. A., Pauchard, A., & Dukes, J. S. (2021). Railways redistribute plant species in mountain landscapes. *Journal of Applied Ecology, 58*(9), 1967–1980. https://doi.org/10.1111/1365-2664.13961

Ratnam, J., Sheth, C., & Sankaran, M. (2019). African and Asian Savannas: Comparisons of Vegetation Composition and Drivers of Vegetation Structure and Function. In P. F. Scogings & M. Sankaran (Eds.), *Savanna Woody Plants and Large Herbivores* (1st ed., pp. 25–49). Wiley. https://doi.org/10.1002/9781119081111.ch2

Raulerson, L. (2006). *Checklist of Plants of the Mariana Islands* (37; pp. 1–69). University of Guam Herbarium Contribution.

Raybaud, V., Beaugrand, G., Dewarumez, J.-M., & Luczak, C. (2015). Climate-induced range shifts of the American jackknife clam *Ensis directus* in Europe. *Biological Invasions, 17*(2), 725–741. https://doi.org/10.1007/s10530-014-0764-4

Reaser, J. K., & Meyerson, L. A. (2003). *Pilot Assessments: The Ecological and Socio-economic Impact of Invasive Alien Species on Island Ecosystems* [Convention on Biological Diversity UNEP/CBD/SBSTAA/9/INF/33]. Convention on Biological Diversity & The Global Invasive Species Programme. https://www.cbd.int/doc/meetings/sbstta/sbstta-09/information/sbstta-09-inf-33-en.pdf

Reaser, J. K., Meyerson, L. A., Cronk, Q., De Poorter, M., Eldrege, L. G., Green, E., Kairo, M., Latasi, P., Mack, R. N., Mauremootoo, J., O'Dowd, D., Orapa, W., Sastroutomo, S., Saunders, A., Shine, C., Thrainsson, S., & Vaiutu, L. (2007). Ecological and socioeconomic impacts of invasive alien species in island ecosystems. *Environmental Conservation, 34*(2), 98–111. https://doi.org/10.1017/S0376892907003815

Rebelo, M. F., Afonso, L. F., Americo, J. A., da Silva, L., Neto, J. L. B., Dondero, F., & Zhang, Q. (2018). A sustainable synthetic biology approach for the control of the invasive golden mussel *(Limnoperna fortunei). PeerJ Preprints, 6*, e27164v3. https://doi.org/10.7287/peerj.preprints.27164v3

Redding, D. W., Pigot, A. L., Dyer, E. E., Şekercioğlu, Ç. H., Kark, S., & Blackburn, T. M. (2019). Location-level processes drive the establishment of alien bird populations worldwide. *Nature, 571*(7763), 103–106. https://doi.org/10.1038/s41586-019-1292-2

Reed, R. N., & Kraus, F. (2010). Invasive reptiles and amphibians: Global perspectives and local solutions. *Animal Conservation, 13*(s1), 3–4. https://doi.org/10.1111/j.1469-1795.2010.00409.x

Reeves, M., Butler, J. L., Schoeneberger, M., & Kilgo, J. C. (2021). Regional summaries: Great Plains Region. In T. M. Poland, T. Patel-Weynand, D. M. Finch, C. Ford Miniat, D. C. Hayes, & V. M. Lopez (Eds.), *Invasive Species in Forests and Rangelands of the United States: A Comprehensive Science Synthesis for the United States Forest Sector* (pp. 402–413). Springer International Publishing. https://doi.org/10.1007/978-3-030-45367-1

Reichard, S. H., & White, P. (2001). Horticulture as a Pathway of Invasive Plant Introductions in the United StatesMost invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. *BioScience, 51*(2), 103–113. https://doi.org/10.1641/0006-3568(2001)051[0103:HAAPOI]2.0.CO;2

Reino, L., Figueira, R., Beja, P., Araújo, M. B., Capinha, C., & Strubbe, D. (2017). Networks of global bird invasion altered by regional trade ban. *Science Advances, 3*(11), e1700783. https://doi.org/10.1126/sciadv.1700783

Reise, K., Gollasch, S., & Wolff, W. J. (1998). Introduced marine species of the North Sea coasts. *Helgolander Meeresunters, 52*(3–4), 219–234. https://doi.org/10.1007/BF02908898

Rejmánek, M., & Randall, J. M. (2004). The total number of naturalized species can be a reliable predictor of the number of alien pest species. *Diversity and Distributions, 10*(5–6), 367–369. https://doi.org/10.1111/j.1366-9516.2004.00107.x

Rejmanek, M., & Richardson, D. M. (1996). What Attributes Make Some Plant Species More Invasive? *Ecology, 77*(6), 1655–1661. https://doi.org/10.2307/2265768

Rejmánek, M., & Richardson, D. M. (2013). Trees and shrubs as invasive alien species—2013 update of the global database. *Diversity and Distributions, 19*(8), 1093–1094. https://doi.org/10.1111/ddi.12075

Rejmánek, M., Richardson, D. M., & Pyšek, P. (2013). Plant Invasions and Invasibility of Plant Communities. In E. van der Maarel & J. Franklin (Eds.), *Vegetation Ecology* (pp. 387–424). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118452592.ch13

Rewicz, T., Grabowski, M., MacNeil, C., & Bącela-Spychalska, K. (2014). The profile of a 'perfect' invader – the case of killer shrimp, *Dikerogammarus villosus. Aquatic Invasions, 9*(3), 267–288. https://doi.org/10.3391/ai.2014.9.3.04

Reynolds, J. W. (2016). Earthworms (Oligochaeta: Acanthodrilidae, Lumbricidae, Megascolecidae and Sparganophilidae) in the Western Corn Belt Ecoregion (47), USA. *Megadrilogica, 21*(10), 190–197.

Reynolds, S. C. P. (2002). *A catalogue of alien plants in Ireland.* National Botanic Gardens.

Ribera, M. A., & Boudouresque, C. F. (1995). Introduced marine plants, with special reference to macroalgae: Mechanisms and impact. *Progress in Phycological Research, 11*, 187–268. https://people.mio.osupytheas.fr/~boudouresque/Publications_pdf/Ribera_Boudouresque_1995_Prog_Phycol_Res.pdf

Ribera Siguan, M. A. (2003). Pathways of biological invasions of marine plants. In G. M. Ruiz & J. T. Carlton (Eds.), *Invasive species: Vectors and management strategies* (pp. 183–226). Island Press. https://www.cabdirect.org/cabdirect/abstract/20043133955

Ricciardi, A. (2001). Facilitative interactions among aquatic invaders: Is an "invasional meltdown" occurring in the Great Lakes? *Canadian Journal of Fisheries and Aquatic Sciences, 58*(12), 2513–2525. https://doi.org/10.1139/f01-178

Ricciardi, A. (2006). Patterns of invasion in the Laurentian Great Lakes in relation to changes in vector activity. *Diversity and Distributions, 12*(4), 425–433. https://doi.org/10.1111/j.1366-9516.2006.00262.x

Ricciardi, A. (2015). Ecology of Invasive Alien Invertebrates. In J. H. Thorp & D. C. Rogers (Eds.), *Ecology and General Biology. Thorp and Covich's Freshwater Invertebrates (4th edn)* (Vol. 1, pp. 83–91). Academic Press, Elsevier Inc. https://doi.org/10.1016/B978-0-12-385026-3.00005-X

Ricciardi, A., Blackburn, T. M., Carlton, J. T., Dick, J. T. A., Hulme, P. E., Iacarella,

J. C., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Sutherland, W. J., Wardle, D. A., & Aldridge, D. C. (2017). Invasion Science: Looking forward rather than revisiting old ground. *Trends in Ecology & Evolution, 32*(11), 809–810. https://doi.org/10.1016/j.tree.2017.08.007

Ricciardi, A., & MacIsaac, H. J. (2000). Recent mass invasion of the North American Great Lakes by Ponto–Caspian species. *Trends in Ecology & Evolution, 15*(2), 62–65. https://doi.org/10.1016/S0169-5347(99)01745-0

Ricciardi, A., & MacIsaac, H. J. (2022). Vector control reduces the rate of species invasion in the world's largest freshwater ecosystem. *Conservation Letters, 15*(2), e12866. https://doi.org/10.1111/conl.12866

Richards, C. L., Bossdorf, O., Muth, N. Z., Gurevitch, J., & Pigliucci, M. (2006). Jack of all trades, master of some? On the role of phenotypic plasticity in plant invasions. *Ecology Letters, 9*(8), 981–993. https://doi.org/10.1111/j.1461-0248.2006.00950.x

Richards, C. L., Schrey, A. W., & Pigliucci, M. (2012). Invasion of diverse habitats by few Japanese knotweed genotypes is correlated with epigenetic differentiation. *Ecology Letters, 15*(9), 1016–1025. https://doi.org/10.1111/j.1461-0248.2012.01824.x

Richardson, A. J., Bakun, A., Hays, G. C., & Gibbons, M. J. (2009). The jellyfish joyride: Causes, consequences and management responses to a more gelatinous future. *Trends in Ecology & Evolution, 24*(6), 312–322. https://doi.org/10.1016/j.tree.2009.01.010

Richardson, D. M. (2006). *Pinus*: A model group for unlocking the secrets of alien plant invasions? *Preslia, 78*, 375–388. https://www.preslia.cz/P064CRic.pdf

Richardson, D. M., Binggeli, P., & Schroth, G. (2004). Invasive Agroforestry trees: Problems and solutions. In G. Schroth, G. A. B. De Fonseca, C. A. Harvey, C. Gascon, H. Vasconcelos, & A.-M. N. Izac (Eds.), *Agroforestry and biodiversity conservation in tropical landscapes* (pp. 371–396). Island Press. https://islandpress.org/books/agroforestry-and-biodiversity-conservation-tropical-landscapes

Richardson, D. M., Carruthers, J., Hui, C., Impson, F. A. C., Miller, J. T., Robertson, M. P., Rouget, M., Le Roux, J. J., & Wilson, J. R. U. (2011). Human-mediated introductions of Australian acacias – a global experiment in biogeography. *Diversity and Distributions, 17*(5), 771–787. https://doi.org/10.1111/j.1472-4642.2011.00824.x

Richardson, D. M., Foxcroft, L. C., Latombe, G., Le Maitre, D. C., Rouget, M., & Wilson, J. R. U. (2020). The biogeography of South African terrestrial plant invasions. In B. W. Van Wilgen, J. Measey, D. M. Richardson, J. R. U. Wilson, & T. A. Zengeya (Eds.), *Biological invasions in South Africa* (Vol. 14, pp. 67–96). Springer. https://doi.org/10.1007/978-3-030-32394-3_3

Richardson, D. M., Holmes, P. M., Esler, K. J., Galatowitsch, S. M., Stromberg, J. C., Kirkman, S. P., Pyšek, P., & Hobbs, R. J. (2007). Riparian vegetation: Degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions, 13*(1), 126–139. https://doi.org/10.1111/j.1366-9516.2006.00314.x

Richardson, D. M., & Rejmánek, M. (2011). Trees and shrubs as invasive alien species–a global review. *Diversity and Distributions, 17*(5), 788–809. https://doi.org/10.1111/j.1472-4642.2011.00782.x

Richardson, D. M., van Wilgen, B. W., & Nuñez, M. A. (2008). Alien conifer invasions in South America: Short fuse burning? *Biological Invasions, 10*(4), 573–577. https://doi.org/10.1007/s10530-007-9140-y

Richardson, D. M., Williams, P. A., & Hobbs, R. J. (1994). Pine invasions in the Southern Hemisphere: Determinants of spread and invadability. *Journal of Biogeography, 21*(5), 511–527. https://doi.org/10.2307/2845655

Richardson, D. M., Witt, A. B. R., Pergl, J., Dawson, W., Essl, F., Kreft, H., van Kleunen, M., Weigelt, P., Winter, M., & Pyšek, P. (2021). Plant invasions in Africa. In D. R. Clements, M. K. Upadhyaya, S. Joshi, & A. Shrestha (Eds.), *Global Plant Invasions* (pp. 225–252). Springer. https://doi.org/10.1007/978-3-030-89684-3_11

Richardson, M. J. (2008). Records of Coprophilous Fungi from the Lesser Antilles and Puerto Rico. *Caribbean Journal of Science, 44*(2), 206–214. https://doi.org/10.18475/cjos.v44i2.a8

Richmond, J. Q., Wood, D. A., Stanford, J. W., & Fisher, R. N. (2015). Testing for multiple invasion routes and source populations for the invasive brown treesnake *(Boiga irregularis)* on Guam: Implications for pest management. *Biological Invasions, 17*(1), 337–349. https://doi.org/10.1007/s10530-014-0733-y

Ricotta, C., La Sorte, F. A., Pyšek, P., Rapson, G. L., Celesti-Grapow, L., & Thompson, K. (2009). Phyloecology of urban alien floras. *Journal of Ecology, 97*(6), 1243–1251. https://doi.org/10.1111/j.1365-2745.2009.01548.x

Ricotta, C., La Sorte, F. A., Pyšek, P., Rapson, G. L., Celesti-Grapow, L., & Thompson, K. (2012). Phylogenetic beta diversity of native and alien species in European urban floras. *Global Ecology and Biogeography, 21*(7), 751–759. https://doi.org/10.1111/j.1466-8238.2011.00715.x

Rights and Resources Initiative. (2015). *Protected Areas and the Land Rights of Indigenous Peoples and Local Communities: Currrent Issues and Future Agenda* (p. 56). Rights and Resources Initiative. http://www.rightsandresources.org/publication/protected-areas-and-the-land-rights-of-indigenous-peoples-and-local-communities-current-issues-and-future-agenda/

Rius, M., & Darling, J. A. (2014). How important is intraspecific genetic admixture to the success of colonising populations? *Trends in Ecology & Evolution, 29*(4), 233–242. https://doi.org/10.1016/j.tree.2014.02.003

Robert, S., Ravigne, V., Zapater, M.-F., Abadie, C., & Carlier, J. (2012). Contrasting introduction scenarios among continents in the worldwide invasion of the banana fungal pathogen *Mycosphaerella Fijiensis. Molecular Ecology, 21*(5), 1098–1114. https://doi.org/10.1111/j.1365-294X.2011.05432.x

Robinson, C. J., & Wallington, T. J. (2012). Boundary work: Engaging knowledge systems in co-management. *Ecology and Society, 17*(2), 16. https://doi.org/10.5751/es-04836-170216

Robinson, T. B., Alexander, M. E., Simon, C. A., Griffiths, C. L., Peters, K., Sibanda, S., Miza, S., Groenewald, B., Majiedt, P., & Sink, K. J. (2016). Lost in translation? Standardising the terminology used in marine invasion biology and updating South *African alien species lists. African Journal of Marine Science, 38*(1), 129–140. https://doi.org/10.2989/1814232X.2016.1163292

Robinson, T. B., Peters, K., & Brooker, B. (2020). Coastal Invasions: The South African Context. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 229–247). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_9

Rocha-Ortega, M., Rodriguez, P., & Córdoba-Aguilar, A. (2021). Geographical, temporal and taxonomic biases in insect GBIF data on biodiversity and extinction. *Ecological Entomology, 46*(4), 718–728. https://doi.org/10.1111/een.13027

Rodda, G. H., Fritts, T. H., & Conry, P. J. (1992). Origin and Population Growth of the Brown Tree Snake, *Boiga irregularis*, on Guam. *Pacific Science, 46*(1), 46–57. https://core.ac.uk/download/pdf/5094285.pdf

Rodríguez-Merino, A., Fernández-Zamudio, R., García-Murillo, P., & Muñoz, J. (2019). Climatic Niche Shift during *Azolla filiculoides* Invasion and Its Potential Distribution under Future Scenarios. *Plants, 8*(10), 424. https://doi.org/10.3390/plants8100424

Roelke, D. L., Barkoh, A., Brooks, B. W., Grover, J. P., Hambright, K. D., LaClaire, J. W., II, Moeller, P. D. R., & Patino, R. (2016). A chronicle of a killer alga in the west: Ecology, assessment, and management of *Prymnesium parvum* blooms. *Hydrobiologia, 764*(1), 29–50. https://doi.org/10.1007/s10750-015-2273-6

Rogers, H. S., Buhle, E. R., HilleRisLambers, J., Fricke, E. C., Miller, R. H., & Tewksbury, J. J. (2017). Effects of an invasive predator cascade to plants via mutualism disruption. *Nature Communications, 8*(1), 14557. https://doi.org/10.1038/ncomms14557

Rojas-Sandoval, J., & Acevedo-Rodríguez, P. (2015). Naturalization and invasion of alien plants in Puerto Rico and the Virgin Islands. *Biological Invasions, 17*(1), 149–163. https://doi.org/10.1007/s10530-014-0712-3

Roll, U., Dayan, T., Simberloff, D., & Mienis, H. K. (2009). Non-indigenous land and freshwater gastropods in Israel. *Biological Invasions, 11*(8), 1963–1972. https://doi.org/10.1007/s10530-008-9373-4

Roman, J., & Darling, J. A. (2007). Paradox lost: Genetic diversity and the success of aquatic invasions. *Trends in Ecology & Evolution, 22*(9), 454–464. https://doi.org/10.1016/j.tree.2007.07.002

Roques, A., Auger-Rozenberg, M.-A., Blackburn, T. M., Garnas, J., Pyšek, P., Rabitsch, W., Richardson, D. M., Wingfield, M. J., Liebhold, A. M., & Duncan, R. P. (2016). Temporal and interspecific variation in rates of spread for insect species invading Europe during the last 200 years. *Biological Invasions, 18*(4), 907–920. https://doi.org/10.1007/s10530-016-1080-y

Rosa, H. H. M. da. (2020). *Patterns of native and aline plant species distribution along roads and trails on an altitudinal gradient of a tropical mountain forest* [Dissertação (mestrado acadêmico), Universidade Federal de Lavras]. http://177.105.2.222/bitstream/1/42381/1/DISSERTA%c3%87%c3%83O_Patterns%20of%20native%20and%20aline%20plant%20species%20distribution%20along%20roads%20and%20trails%20on%20an%20altitudinal%20gradient%20of%20a%20tropical%20mountain%20forest.pdf

Rossiter, N. A., Setterfield, S. A., Douglas, M. M., & Hutley, L. B. (2003). Testing the grass-fire cycle: Alien grass invasion in the tropical savannas of northern Australia. *Diversity and Distributions, 9*(3), 169–176. https://doi.org/10.1046/j.1472-4642.2003.00020.x

Roué, M., Césard, N., Adou Yao, Y. C., & Oteng-Yeboah, A. (Eds.). (2017). *Knowing our lands and resources: Indigenous and local knowledge of biodiversity and ecosystem services in Africa.* Knowledges of Nature 8. UNESCO: Paris. https://unesdoc.unesco.org/ark:/48223/pf0000247461

Roura-Pascual, N., Leung, B., Rabitsch, W., Rutting, L., Vervoort, J., Bacher, S., Dullinger, S., Erb, K.-H., Jeschke, J. M., Katsanevakis, S., Kühn, I., Lenzner, B., Liebhold, A. M., Obersteiner, M., Pauchard, A., Peterson, G. D., Roy, H. E., Seebens, H., Winter, M., … Essl, F. (2021). Alternative futures for global biological invasions. *Sustainability Science, 16*(5), 1637–1650. https://doi.org/10.1007/s11625-021-00963-6

Roy, H. E., Adriaens, T., Isaac, N. J. B., Kenis, M., Onkelinx, T., Martin, G. S., Brown, P. M. J., Hautier, L., Poland, R., Roy, D. B., Comont, R., Eschen, R., Frost, R., Zindel, R., Van Vlaenderen, J., Nedvěd, O., Ravn, H. P., Grégoire, J.-C., de Biseau, J.-C., & Maes, D. (2012). Invasive alien predator causes rapid declines of native European ladybirds. *Diversity and Distributions, 18*(7), 717–725. https://doi.org/10.1111/j.1472-4642.2012.00883.x

Roy, H. E., Hesketh, H., Purse, B. V., Eilenberg, J., Santini, A., Scalera, R., Stentiford, G. D., Adriaens, T., Bacela-Spychalska, K., Bass, D., Beckmann, K. M., Bessell, P., Bojko, J., Booy, O., Cardoso, A. C., Essl, F., Groom, Q., Harrower, C., Kleespies, R., … Dunn, A. M. (2017). Alien Pathogens on the Horizon: Opportunities for Predicting their Threat to Wildlife. *Conservation Letters, 10*(4), 477–484. https://doi.org/10.1111/conl.12297

Roy, H. E., Peyton, J. M., Aldridge, D. C., Bantock, T., Blackburn, T. M., Britton, R., Clark, P., Cook, E., Dehnen-Schmutz, K., Dines, T., Dobson, M., Edwards, F., Harrower, C., Harvey, M. C., Minchin, D., Noble, D. G., Parrott, D., Pocock, M. J. O., Preston, C. D., … Walker, K. J. (2014). Horizon scanning for invasive alien species with the potential to threaten biodiversity in Great Britain. *Global Change Biology, 20*(12), 3859–3871. https://doi.org/10.1111/gcb.12603

Roy, H. E., Preston, C. D., Harrower, C. A., Rorke, S. L., Noble, D., Sewell, J., Walker, K., Marchant, J., Seeley, B., Bishop, J., Jukes, A., Musgrove, A., Pearman, D., & Booy, O. (2014). GB Non-native Species Information Portal: Documenting the arrival of non-native species in Britain. *Biological Invasions, 16*(12), 2495–2505. https://doi.org/10.1007/s10530-014-0687-0

Roy, H. E., Rabitsch, W., Scalera, R., Stewart, A., Gallardo, B., Genovesi, P., Essl, F., Adriaens, T., Bacher, S., Booy, O., Branquart, E., Brunel, S., Copp, G. H., Dean, H., D'hondt, B., Josefsson, M., Kenis, M., Kettunen, M., Linnamagi, M., … Zenetos, A. (2018). Developing a framework of minimum standards for the risk assessment of alien species. *Journal of Applied Ecology, 55*(2), 526–538. https://doi.org/10.1111/1365-2664.13025

Rozzi, R., Armesto, J. J., Goffinet, B., Buck, W., Massardo, F., Silander, J., Arroyo, M. T., Russell, S., Anderson, C. B., Cavieres, L. A., & Callicott, J. B. (2008). Changing lenses to assess biodiversity: Patterns of species richness in sub-Antarctic plants and implications for global conservation. *Frontiers in Ecology and the Environment, 6*(3), 131–137. https://doi.org/10.1890/070020

Rúa, M. A., Pollina, E. C., Power, A. G., & Mitchell, C. E. (2011). The role of viruses in biological invasions: Friend or foe? *Current Opinion in Virology, 1*(1), 68–72. https://doi.org/10.1016/j.coviro.2011.05.018

Ruesink, J. L., Lenihan, H. S., Trimble, A. C., Heiman, K. W., Micheli, F., Byers, J. E., & Kay, M. C. (2005). Introduction of Non-Native Oysters: Ecosystem Effects and Restoration Implications. *Annual Review of Ecology, Evolution, and Systematics, 36*(1), 643–689. https://doi.org/10.1146/annurev.ecolsys.36.102003.152638

Ruffino, L., Bourgeois, K., Vidal, E., Duhem, C., Paracuellos, M., Escribano, F., Sposimo, P., Baccetti, N., Pascal, M., & Oro, D. (2009). Invasive rats and seabirds after 2,000 years of an unwanted coexistence on Mediterranean islands. *Biological Invasions,* *11*(7), 1631–1651. https://doi.org/10.1007/s10530-008-9394-z

Ruiz, G. M., & Carlton, J. T. (Eds.). (2003). *Invasive Species: Vectors And Management Strategies.* Island Press.

Ruiz, G. M., Carlton, J. T., Grosholz, E. D., & Hines, A. H. (1997). Global invasions of marine and estuarine habitats by non-indigenous species: Mechanisms, extent and consequences. *American Zoologist, 37*(6), 621–632. https://doi.org/10.1093/icb/37.6.621

Ruiz, G. M., Fofonoff, P., Steves, B., & Dahlstrom, A. (2011). Marine Crustacean Invasions in North America: A Synthesis of Historical Records and Documented Impacts. In B. S. Galil, P. F. Clark, & J. T. Carlton (Eds.), *In the Wrong Place—Alien Marine Crustaceans: Distribution, Biology and Impacts* (Vol. 6, pp. 215–250). Springer Netherlands. https://doi.org/10.1007/978-94-007-0591-3_6

Ruiz, G. M., Fofonoff, P. W., Carlton, J. T., Wonham, M. J., & Hines, A. H. (2000). Invasion of coastal marine communities in North America: Apparent patterns, processes, and biases. *Annual Review of Ecology and Systematics, 31*(1), 481–531. https://doi.org/10.1146/annurev.ecolsys.31.1.481

Ruiz, G. M., Fofonoff, P. W., Steves, B. P., & Carlton, J. T. (2015). Invasion history and vector dynamics in coastal marine ecosystems: A North American perspective. *Aquatic Ecosystem Health and Management, 18*(3), 299–311. https://doi.org/10.1080/14634988.2015.1027534

Ruiz, G. M., & Hewitt, C. L. (2009). Latitudinal patterns of biological invasions in marine ecosystems: A polar perspective. In I. Krupnik, M. A. Lang, & S. E. Miller (Eds.), Smithsonian at the poles: Contributions to International Polar Year science (pp. 347–358). *Smithsonian Institution Scholarly Press*. https://doi.org/10.5479/si.097884601X.26

Ruiz, G. M., Lorda, J., Arnwine, A., & Lion, K. (2006). Chapter II Shipping Patterns Associated with the Panama Canal: Effects on Biotic Exchange? In S. Gollasch, B. S. Galil, & A. N. Cohen (Eds.), *Bridging Divides: Maritime Canals as Invasion Corridors* (Vol. 83, pp. 113–126). Springer Netherlands. https://doi.org/10.1007/978-1-4020-5047-3_4

Ruiz, G. M., Rawlings, T. K., Dobbs, F. C., Drake, L. A., Mullady, T., Huq, A., & Colwell, R. R. (2000). Global spread of microorganisms by ships—Ballast water discharged from vessels harbours a cocktail of potential pathogens. *Nature, 408*(6808), 49–50. https://doi.org/10.1038/35040695

Russell, J. C., Cole, N. C., Zuël, N., & Rocamora, G. (2016). Introduced mammals on Western Indian Ocean islands. *Global Ecology and Conservation, 6*, 132–144. https://doi.org/10.1016/j.gecco.2016.02.005

Russell, J. C., Innes, J. G., Brown, P. H., & Byrom, A. E. (2015). Predator-Free New Zealand: Conservation Country. *BioScience, 65*(5), 520–525. https://doi.org/10.1093/biosci/biv012

Russell, J. C., & Kueffer, C. (2019). Island Biodiversity in the Anthropocene. *Annual Review of Environment and Resources, 44*(1), 31–60. https://doi.org/10.1146/annurev-environ-101718-033245

Russell, J. C., & Le Corre, M. (2009). Introduced mammal impacts on seabirds in the Îles Éparses, Western Indian Ocean. *Marine Ornithology, 37*, 121–128.

Russell, J. C., Meyer, J.-Y., Holmes, N. D., & Pagad, S. (2017). Invasive alien species on islands: Impacts, distribution, interactions and management. *Environmental Conservation, 44*(4), 359–370. https://doi.org/10.1017/S0376892917000297

Ruttenberg, B. I., Schofield, P. J., Akins, J. L., Acosta, A., Feeley, M. W., Blondeau, J., Smith, S. G., & Ault, J. S. (2012). Rapid invasion of the Indo-Pacific lionfishes (*Pterois volitans* and *Pterois miles*) in the Florida Keys, USA: evidence from multiple pre-and post-invasion data sets. *Bulletin of Marine Science,* 88(4), 10511059. https://doi.org/10.5343/bms.2011.1108

Ryan, S. F., Lombaert, E., Espeset, A., Vila, R., Talavera, G., Dinca, V., Doellman, M. M., Renshaw, M. A., Eng, M. W., Hornett, E. A., Li, Y., Pfrender, M. E., & Shoemaker, D. (2019). Global invasion history of the agricultural pest butterfly *Pieris rapae* revealed with genomics and citizen science. *Proceedings of the National Academy of Sciences of the United States of America, 116*(40), 20015–20024. https://doi.org/10.1073/pnas.1907492116

Ryan, S. J., Carlson, C. J., Mordecai, E. A., & Johnson, L. R. (2019). Global expansion and redistribution of *Aedes*-borne virus transmission risk with climate change. *PLoS Neglected Tropical Diseases, 13*(3), e0007213. https://doi.org/10.1371/journal.pntd.0007213

Saba, A. O., Ismail, A., Zulkifli, S. Z., Shohaimi, S., Jamil, N. R., Nawi, N. M., Ghani, I. F. A., Halim, M. R. A., & Amal, M. N. A. (2020). Checklists, production trends, and potential ecological and socioeconomic impacts of non-native freshwater fishes in Malaysia: A review. *Aquatic Invasions, 15*(4), 646–670. https://doi.org/10.3391/ai.2020.15.4.07

Saeedi, H., Bernardino, A. F., Shimabukuro, M., Falchetto, G., & Sumida, P. Y. G. (2019). Macrofaunal community structure and biodiversity patterns based on a wood-fall experiment in the deep South-west Atlantic. *Deep Sea Research Part I: Oceanographic Research Papers, 145*, 73–82. https://doi.org/10.1016/j.dsr.2019.01.008

Saeedi, H., & Brandt, A. (2020). Biogeographic Atlas of the Deep NW Pacific Fauna. *Advanced Books, 1*, e51315. https://doi.org/10.3897/ab.e51315

Saeedi, H., Costello, M. J., Warren, D., & Brandt, A. (2019). Latitudinal and bathymetrical species richness patterns in the NW Pacific and adjacent Arctic Ocean. *Scientific Reports, 9*(1), 9303. https://doi.org/10.1038/s41598-019-45813-9

Saeedi, H., Simões, M., & Brandt, A. (2020). Biodiversity and distribution patterns of deep-sea fauna along the temperate NW Pacific. *Progress in Oceanography, 183*, 102296. https://doi.org/10.1016/j.pocean.2020.102296

Sailer, R. I. (1983). History of Insect Introductions. In C. L. Wilson & C. L. Graham (Eds.), *Exotic Plant Pests and North American Agriculture* (pp. 15–38). Elsevier. https://doi.org/10.1016/B978-0-12-757880-4.50007-5

Sakai, A. K., Allendorf, F. W., Holt, J. S., Lodge, D. M., Molofsky, J., With, K. A., Baughman, S., Cabin, R. J., Cohen, J. E., Ellstrand, N. C., McCauley, D. E., O'Neil, P., Parker, I. M., Thompson, J. N., & Weller, S. G. (2001). The population biology of invasive species. *Annual Review of Ecology and Systematics, 32*(1), 305–332. https://doi.org/10.1146/annurev.ecolsys.32.081501.114037

Sanchéz, A. L. (1997). Los jardines botánicos neotropciales y el intercambio de plantas: Pasado, presente y futuro. *Monografías Del Real Jardín Botánico de Córdoba, 5*, 75–84.

Sanderson, L. A., Mclaughlin, J. A., & Antunes, P. M. (2012). The last great forest: A review of the status of invasive species in the North American boreal forest. *Forestry, 85*(3), 329–340. https://doi.org/10.1093/forestry/cps033

Sandquist, D. R. (2014). Plants in Deserts. In R. K. Monson (Ed.), *Ecology and the Environment* (Vol. 8, pp. 297–326). Springer New York. https://doi.org/10.1007/978-1-4614-7501-9_3

Sandvik, H., Dolmen, D., Elven, R., Falkenhaug, T., Forsgren, E., Hansen, H., Hassel, K., Husa, V., Kjærstad, G., Ødegaard, F., Pedersen, H. C., Solheim, H., Stokke, B. G., Åsen, P. A., Åström, S., Brandrud, T.-E., Elven, H., Endrestøl, A., Finstad, A., … Gederaas, L. (2019). Alien plants, animals, fungi and algae in Norway: An inventory of neobiota. *Biological Invasions, 21*(10), 2997–3012. https://doi.org/10.1007/s10530-019-02058-x

Sandvik, H., Hilmo, O., Finstad, A. G., Hegre, H., Moen, T. L., Rafoss, T., Skarpaas, O., Elven, R., Sandmark, H., & Gederaas, L. (2019). Generic ecological impact assessment of alien species (GEIAA): The third generation of assessments in Norway. *Biological Invasions, 21*(9), 2803–2810. https://doi.org/10.1007/s10530-019-02033-6

Santelices, B. (1989). *Algas marinas de Chile: Distribución, ecología, utilización y diversidades.* Ediciones Universidad Católica de Chile. https://books.google.cl/books/about/Algas_marinas_de_Chile.html?id=Gwf7nAEACAAJ&redir_esc=y

Santini, A., Ghelardini, L., De Pace, C., Desprez-Loustau, M. L., Capretti, P., Chandelier, A., Cech, T., Chira, D., Diamandis, S., Gaitniekis, T., Hantula, J., Holdenrieder, O., Jankovsky, L., Jung, T., Jurc, D., Kirisits, T., Kunca, A., Lygis, V., Malecka, M., … Stenlid, J. (2013). Biogeographical patterns and determinants of invasion by forest pathogens in Europe. *New Phytologist, 197*(1), 238–250. https://doi.org/10.1111/j.1469-8137.2012.04364.x

Sardain, A., Sardain, E., & Leung, B. (2019). Global forecasts of shipping traffic and biological invasions to 2050. *Nature Sustainability, 2*(4), 274–282. https://doi.org/10.1038/s41893-019-0245-y

Satia, B. P., & Bartley, D. M. (1998). The paradox of international introductions of aquatic organisms in Africa. In J. F. Agnèse (Ed.), *Genetics and aquaculture in Africa* (pp. 115–124). Éditions de l'Orstom.

Saul, W.-C., Roy, H. E., Booy, O., Carnevali, L., Chen, H.-J., Genovesi, P., Harrower, C. A., Hulme, P. E., Pagad, S., Pergl, J., & Jeschke, J. M. (2017). Assessing patterns in introduction pathways of alien species by linking major invasion data bases. *Journal of Applied Ecology, 54*(2), 657–669. https://doi.org/10.1111/1365-2664.12819

Savini, D., Occhipinti–Ambrogi, A., Marchini, A., Tricarico, E., Gherardi, F., Olenin, S., & Gollasch, S. (2010). The top 27 animal alien species introduced into Europe for aquaculture and related activities. *Journal of Applied Ichthyology, 26*(s2), 1–7. https://doi.org/10.1111/j.1439-0426.2010.01503.x

Sax, D. F. (2001). Latitudinal gradients and geographic ranges of exotic species: Implications for biogeography. *Journal of Biogeography, 28*(1), 139–150. https://doi.org/10.1046/j.1365-2699.2001.00536.x

Sax, D. F., & Gaines, S. D. (2008). Species invasions and extinction: The future of native biodiversity on islands. *Proceedings of the National Academy of Sciences, 105*(Supplement 1), 11490–11497. https://doi.org/10.1073/pnas.0802290105

Schlichting, C. D. (1986). The evolution of phenotypic plasticity in plants. *Annual Review of Ecology and Systematics, 17*(1), 667–693. https://doi.org/10.1146/annurev.es.17.110186.003315

Schmidt, A. L., & Scheibling, R. E. (2006). A comparison of epifauna and epiphytes on native kelps *(Laminaria species)* and an invasive alga (*Codium fragile* ssp. *Tomentosoides*) in Nova Scotia, Canada. *Botanica Marina, 49*(4), 315–330. https://doi.org/10.1515/BOT.2006.039

Schöll, F. (2007). Der Main-Donau-Kanal als Drehscheibe wandernder Tierarten. In *Wasser- und Schifffahrtsdirektion Süd (Hrsg.)* (pp. 69–70). Jahresbericht.

Schulze, K., Knights, K., Coad, L., Geldmann, J., Leverington, F., Eassom, A., Marr, M., Butchart, S. H. M., Hockings, M., & Burgess, N. D. (2018). An assessment of threats to terrestrial protected areas. *Conservation Letters, 11*(3), e12435. https://doi.org/10.1111/conl.12435

Schwindt, E., Battini, N., Giachetti, C., Castro, K., & Bortolus, A. (2018). *Especies exóticas marino-costeras: Argentina = Marine-coastal exotic species: Argentina* (pp. 1–170). Vazquez Mazzini Editores. https://www.biodiversitylibrary.org/item/281334

Schwindt, E., & Bortolus, A. (2017). Aquatic invasion biology research in South America: Geographic patterns, advances and perspectives. *Aquatic Ecosystem Health & Management, 20*(4), 322–333. https://doi.org/10.1080/14634988.2017.1404413

Schwindt, E., Carlton, J. T., Orensanz, J. M., Scarabino, F., & Bortolus, A. (2020). Past and future of the marine bioinvasions along the Southwestern Atlantic. *Aquatic Invasions, 15*(1), 11–29. https://doi.org/10.3391/ai.2020.15.1.02

Scoppettone, G. G., Rissler, P. H., Gourley, C., & Martinez, C. (2005). Habitat restoration as a means of controlling non-native fish in a Mojave desert Oasis. *Restoration Ecology, 13*(2), 247–256. https://doi.org/10.1111/j.1526-100X.2005.00032.x

Scott, P., Bader, M. K.-F., Burgess, T., Hardy, G., & Williams, N. (2019). Global biogeography and invasion risk of the plant pathogen genus *Phytophthora. Environmental Science & Policy, 101*, 175–182. https://doi.org/10.1016/j.envsci.2019.08.020

Seastedt, T. R., & Pyšek, P. (2011). Mechanisms of plant invasions of North American and European grasslands. *Annual Review of Ecology, Evolution, and Systematics, 42*(1), 133–153. https://doi.org/10.1146/annurev-ecolsys-102710-145057

Seaward, K., & Inglis, G. J. (2018). *Long-term indicators for non-indigenous species (NIS) in marine systems* (NIWA Client Report No: 2018310CH; p. 74). The Ministry for the Environment, New Zealand Government. https://environment.govt.nz/assets/Publications/Files/long-term-indicators-for-non-indigenous-species-in-marine-systems.pdf

Secretariat of the CBD. (2014). *Global Biodiversity Outlook 4* (p. 155). Secretariat of the Convention on Biological Diversity. https://www.cbd.int/gbo/gbo4/publication/gbo4-en-hr.pdf

Secretariat of the CBD. (2020). *Global Biodiversity Outlook 5* (p. 211). Secretariat of the Convention on Biological Diversity. https://www.cbd.int/gbo/gbo5/publication/gbo-5-en.pdf

Seddon, P. J., Moro, D., Mitchell, N. J., Chauvenet, A. L. M., & Mawson, P. R. (2015). Proactive conservation or planned invasions? Past, current and future use of assisted colonisation. In D. P. Armstrong, M. W. Hayward, D. Moro, & P. J. Seddon (Eds.), *Advances in Reintroduction Biology of Australian and New Zealand Fauna* (pp. 105–125). CSIRO Australia. http://www.publish.csiro.au/book/7357/

Seebens, H. (2021). *SInAS database of alien species occurrences* (2.4.1) [dataset]. Zenodo. https://doi.org/10.5281/ZENODO.5562892

Seebens, H., Bacher, S., Blackburn, T. M., Capinha, C., Dawson, W., Dullinger, S., Genovesi, P., Hulme, P. E., Kleunen, M., Kühn, I., Jeschke, J. M., Lenzner, B., Liebhold, A. M., Pattison, Z., Pergl, J., Pyšek, P., Winter, M., & Essl, F. (2021). Projecting the continental accumulation of alien species through to 2050. *Global Change Biology, 27*(5), 970–982. https://doi.org/10.1111/gcb.15333

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., van Kleunen, M., Winter, M., Ansong, M., Arianoutsou, M., Bacher, S., Blasius, B., Brockerhoff, E. G., Brundu, G., Capinha, C., Causton, C. E., Celesti-Grapow, L., … Essl, F. (2018). Global rise in emerging alien species results from increased accessibility of new source pools. *Proceedings of the National Academy of Sciences, 115*(10), E2264–E2273. https://doi.org/10.1073/pnas.1719429115

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., … Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications, 8*(1), 14435. https://doi.org/10.1038/ncomms14435

Seebens, H., Blackburn, T. M., Hulme, P. E., Van Kleunen, M., Liebhold, A. M., Orlova-Bienkowskaja, M., Pyšek, P., Schindler, S., & Essl, F. (2021). Around the world in 500 years: Inter-regional spread of alien species over recent centuries. *Global Ecology and Biogeography, 30*(8), 1621–1632. https://doi.org/10.1111/geb.13325

Seebens, H., Clarke, D. A., Groom, Q., Wilson, J. R. U., García-Berthou, E., Kühn, I., Roigé, M., Pagad, S., Essl, F., Vicente, J., Winter, M., & McGeoch, M. A. (2020). A workflow for standardising and integrating alien species distribution data. *NeoBiota, 59*, 39–59. https://doi.org/10.3897/neobiota.59.53578

Seebens, H., Essl, F., & Blasius, B. (2017). The intermediate distance hypothesis of biological invasions. *Ecology Letters, 20*(2), 158–165. https://doi.org/10.1111/ele.12715

Seebens, H., Essl, F., Dawson, W., Fuentes, N., Moser, D., Pergl, J., Pyšek, P., Van Kleunen, M., Weber, E., Winter, M., & Blasius, B. (2015). Global trade will accelerate plant invasions in emerging economies under climate change. *Global Change Biology, 21*(11), 4128–4140. https://doi.org/10.1111/gcb.13021

Seebens, H., Gastner, M. T., & Blasius, B. (2013). The risk of marine bioinvasion caused by global shipping. *Ecology Letters, 16*(6), 782–790. https://doi.org/10.1111/ele.12111

Seebens, H., & Kaplan, E. (2022a). *Coordinates of alien populations as obtained from the DASCO workflow and SInAS data 2.4.1* (1.0.1) [dataset]. Zenodo. https://doi.org/10.5281/ZENODO.6458083

Seebens, H., & Kaplan, E. (2022b). DASCO: A workflow to downscale alien species checklists using occurrence records and to re-allocate species distributions across realms. *NeoBiota, 74*, 75–91. https://doi.org/10.3897/neobiota.74.81082

Seebens, H., Schwartz, N., Schupp, P. J., & Blasius, B. (2016). Predicting the spread of marine species introduced by global shipping. *Proceedings of the National Academy of Sciences, 113*(20), 5646–5651. https://doi.org/10.1073/pnas.1524427113

Seipel, T., Kueffer, C., Rew, L. J., Daehler, C. C., Pauchard, A., Naylor, B. J., Alexander, J. M., Edwards, P. J., Parks, C. G., Arevalo, J. R., Cavieres, L. A., Dietz, H., Jakobs, G., McDougall, K., Otto, R., & Walsh, N. (2012). Processes at multiple scales affect richness and similarity of non-native plant species in mountains around the world. *Global Ecology and Biogeography, 21*(2), 236–246. https://doi.org/10.1111/j.1466-8238.2011.00664.x

Senan, A. S., Tomasetto, F., Farcomeni, A., Somashekar, R. K., & Attorre, F. (2012). Determinants of plant species invasions in an arid island: Evidence from Socotra Island (Yemen). *Plant Ecology, 213*(9), 1381–1392. https://doi.org/10.1007/s11258-012-0098-1

Seo, K. S., & Lee, Y. (2009). A First Assessment of Invasive Marine Species on Chinese and Korean Coasts. In G. Rilov & J. A. Crooks (Eds.), *Biological Invasions in Marine Ecosystems* (Vol. 204, pp. 577–585). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-540-79236-9_32

Sepúlveda, M., Arismendi, I., Soto, D., Jara, F., & Farias, F. (2013). Escaped farmed salmon and trout in Chile: Incidence, impacts, and the need for an ecosystem view. *Aquaculture Environment Interactions, 4*(3), 273–283. https://doi.org/10.3354/aei00089

Setterfield, S. A., Douglas, M. M., Petty, A. M., Bayliss, P., Ferdinands, K. B., & Winderlich, S. (2013). Invasive Plants in the Floodplains of Australia's Kakadu National Park. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas. Patterns, Problems and Challenges*. (Vol. 7, pp. 167–189). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_9

Shaanker, R. U., & Ganeshaiah, K. N. (1992). Biological control programmes—Time to retrospect. *Current Science, 63*(12), 732–734. https://www.jstor.org/stable/24095504

Shackleton, R. T., Bertzky, B., Wood, L. E., Bunbury, N., Jäger, H., van Merm, R., Sevilla, C., Smith, K., Wilson, J. R. U., Witt, A. B. R., & Richardson, D. M. (2020). Biological invasions in World Heritage Sites: Current status and a proposed monitoring and reporting framework. *Biodiversity and Conservation, 29*(11), 3327–3347. https://doi.org/10.1007/s10531-020-02026-1

Shackleton, R. T., Biggs, R., Richardson, D. M., & Larson, B. M. H. (2018). Social-ecological drivers and impacts of invasion-related regime shifts: Consequences for ecosystem services and human wellbeing. *Environmental Science & Policy, 89*, 300–314. https://doi.org/10.1016/j.envsci.2018.08.005

Shackleton, R. T., Foxcroft, L. C., Pyšek, P., Wood, L. E., & Richardson, D. M. (2020). Assessing biological invasions in protected areas after 30 years: Revisiting nature reserves targeted by the 1980s SCOPE programme. *Biological Conservation, 243*, 108424. https://doi.org/10.1016/j.biocon.2020.108424

Shackleton, R. T., Le Maitre, D. C., Pasiecznik, N. M., & Richardson, D. M. (2014). *Prosopis*: A global assessment of the biogeography, benefits, impacts and management of one of the world's worst woody invasive plant taxa. *AoB PLANTS, 6*, plu027. https://doi.org/10.1093/aobpla/plu027

Shackleton, S. E., & Shackleton, R. T. (2018). Local knowledge regarding ecosystem services and disservices from invasive alien plants in the arid Kalahari, South Africa. *Journal of Arid Environments, 159*, 22–33. https://doi.org/10.1016/j.jaridenv.2017.07.001

Shafer, D. J., Kaldy, J. E., & Gaeckle, J. L. (2014). Science and management of the introduced seagrass *Zostera japonica* in North America. *Environmental Management, 53*(1), 147–162. https://doi.org/10.1007/s00267-013-0172-z

Shafia, A., & Saleem, A. (2003). Maldives. In N. Pallewatta, J. K. Reaser, & A. T. Gutierrez (Eds.), *Invasive Alien Species in South-Southeast Asia* (p. 111). National Reports & Directory of Resources. Global Invasive Species Programme. https://www.doi.gov/sites/doi.gov/files/uploads/invasive_alien_species_in_south_southeast_asia.pdf

Shaltout, K. H., Hosni, H. A., El-Kady, H. F., El-Beheiry, M. A., & Shaltout, S. K. (2016). Composition and pattern of alien species in the Egyptian flora. *Flora – Morphology, Distribution, Functional Ecology of Plants, 222*, 104–110. https://doi.org/10.1016/j.flora.2016.04.001

Shanmuganathan, T., Pallister, J., Doody, S., McCallum, H., Robinson, T., Sheppard, A., Hardy, C., Halliday, D., Venables, D., Voysey, R., Strive, T., Hinds, L., & Hyatt, A. (2010). Biological control of the cane toad in Australia: A review. *Animal Conservation, 13*(s1), 16–23. https://doi.org/10.1111/j.1469-1795.2009.00319.x

Sharma, R., Polkade, A. V., & Shouche, Y. S. (2015). 'Species concept' in microbial taxonomy and systematics. *Current Science, 108*(10), 1804–1814. https://www.jstor.org/stable/24905604

Shaw, J. D. (2013). Southern Ocean Islands invaded: Conserving biodiversity in the world's last wilderness. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7, pp. 449–470). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_20

Shelton, W. L., & Rothbard, S. (2006). Exotic species in global aquaculture—A review. *Israeli Journal of Aquaculture – Bamidgeh, 58*(1), 3–28. https://doi.org/10.46989/001c.20429

Sheppard, A. W., Shaw, R. H., & Sforza, R. (2006). Top 20 environmental weeds for classical biological control in Europe: A review of opportunities, regulations and other barriers to adoption. *Weed Research, 46*(2), 93–117. https://doi.org/10.1111/j.1365-3180.2006.00497.x

Shi, H. T., Gong, S. P., Liang, W., Hong, M. L., Fu, L. R., & Wang, J. C. (2009). Controlling the trend of alien red-eared sliders *(Trachemys scripta elegans)* spread in the wild of China. *Biol Bull US, 44*, 1–3.

Shiferaw, H., Alamirew, T., Dzikiti, S., Bewket, W., Zeleke, G., & Schaffner, U. (2021). Water use of *Prosopis juliflora* and its impacts on catchment water budget and rural livelihoods in Afar Region, Ethiopia. *Scientific Reports, 11*(1), 2688. https://doi.org/10.1038/s41598-021-81776-6

Shiganova, T. A., Sommer, U., Javidpour, J., Molinero, J. C., Malej, A., Kazmin, A. S., Isinibilir, M., Christou, E., Siokou-Frangou, I., Marambio, M., Fuentes, V., Mirsoyan, Z. A., Gülsahin, N., Lombard, F., Lilley, M. K. S., Angel, D. L., Galil, B. S., Bonnet, D., & Delpy, F. (2019). Patterns of invasive ctenophore *Mnemiopsis leidyi* distribution and variability in different recipient environments of the Eurasian seas: A review. *Marine Environmental Research, 152*, 104791. https://doi.org/10.1016/j.marenvres.2019.104791

Shine, R. (2018). *Cane toad wars*. University of California Press.

Shrestha, B. B. (2016). Invasive alien plant species in Nepal. In P. K. Jha, M. Siwakoti, & S. Rajbhandary (Eds.), *Frontiers of Botany* (pp. 269–284). Central Department of Botany, Tribhuvan University. http://www.invasiveplantsnepal.org/wp-content/uploads/2017/06/2016_Shrestha-BB_Invasive-alien-plant-species-in-Nepal.pdf

Shrestha, B. B., & Shrestha, K. K. (2021). Invasions of alien plant species in Nepal: Patterns and process. In T. Pullaiah & M. R. Ielmini (Eds.), *Invasive Alien Species* (1st ed., pp. 168–183). Wiley. https://doi.org/10.1002/9781119607045.ch20

Sikes, B. A., Bufford, J. L., Hulme, P. E., Cooper, J. A., Johnston, P. R., & Duncan, R. P. (2018). Import volumes and biosecurity interventions shape the arrival rate of fungal pathogens. *PLoS Biology, 16*(5), e2006025. https://doi.org/10.1371/journal.pbio.2006025

Silva-Rocha, I. R., Salvi, D., & Carretero, M. A. (2012). Genetic data reveal a multiple origin for the populations of the Italian wall lizard *Podarcis sicula* (Squamata: Lacertidae) introduced in the Iberian Peninsula and Balearic islands. *Italian Journal of Zoology, 79*(4), 502–510. https://doi.org/10.1080/11250003.2012.680983

Silva-Rocha, I. R., Salvi, D., Carretero, M. A., & Ficetola, G. F. (2019). Alien reptiles on Mediterranean Islands: A model for invasion biogeography. *Diversity and Distributions, 25*(6), 995–1005. https://doi.org/10.1111/ddi.12911

Silva-Rocha, I. R., Salvi, D., Sillero, N., Mateo, J. A., & Carretero, M. A. (2015). Snakes on the Balearic Islands: An invasion tale with implications for native biodiversity conservation. *PLoS ONE, 10*(4), e0121026. https://doi.org/10.1371/journal.pone.0121026

Simberloff, D. (2012). Risks of biological control for conservation purposes. *BioControl, 57*(2), 263–276. https://doi.org/10.1007/s10526-011-9392-4

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vilà, M. (2013). Impacts of biological invasions: What's what and the way forward. *Trends in Ecology & Evolution, 28*(1), 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Simberloff, D., Nuñez, M. A., Ledgard, N. J., Pauchard, A., Richardson, D. M., Sarasola, M., Van Wilgen, B. W., Zalba, S. M., Zenni, R. D., Bustamante, R., Peña, E., & Ziller, S. R. (2009). Spread and impact of introduced conifers in South America: Lessons from other southern hemisphere regions. *Austral Ecology, 35*(5), 489–504. https://doi.org/10.1111/j.1442-9993.2009.02058.x

Simberloff, D., & Rejmanek, M. (Eds.). (2011). *Encyclopedia of Biological Invasions* (Vol. 3). University of California Press. https://doi.org/10.1525/9780520948433

Šimková, A., Řehulková, E., Rasoloariniaina, J. R., Jorissen, M. W. P., Scholz, T., Faltýnková, A., Mašová, Š., & Vanhove, M. P. M. (2019). Transmission of parasites from introduced tilapias: A new threat to endemic Malagasy ichthyofauna. *Biological Invasions, 21*(3), 803–819. https://doi.org/10.1007/s10530-018-1859-0

Simmonds, F. J., Franz, J. M., & Sailer, R. I. (1976). History of biological control. In C. B. Huffaker & P. S. Messenger (Eds.), *Theory and Practice of Biological Control* (pp. 17–39). Academic Press. https://www.cabdirect.org/cabdirect/abstract/19772317404

Simmons, M. T., Windhager, S., Power, P., Lott, J., Lyons, R. K., & Schwope, C. (2007). Selective and non-selective control of invasive plants: The short-term effects of growing-season prescribed fire, herbicide, and mowing in two Texas prairies. *Restoration Ecology, 15*(4), 662–669. https://doi.org/10.1111/j.1526-100X.2007.00278.x

Simões, M. V. P., Saeedi, H., Cobos, M. E., & Brandt, A. (2021). Environmental matching reveals non-uniform range-shift patterns in benthic marine Crustacea. *Climatic Change, 168*, 31. https://doi.org/10.1007/s10584-021-03240-8

Singh, R. P., Hodson, D. P., Huerta-Espino, J., Jin, Y., Njau, P., Wanyera, R., Herrera-Foessel, S. A., & Ward, R. W. (2008). Will Stem Rust Destroy the World's Wheat Crop? In *Advances in Agronomy* (Vol. 98, pp. 271–309). Elsevier. https://doi.org/10.1016/S0065-2113(08)00205-8

Skowno, A., Poole, C. J., Raimondo, D., Sink, K. J., Van Deventer, H., Van Niekerk, L., Harris, L., Smith-Adao, L., Tolley, K., Zengeya, T., Foden, W., Midgley, G., &

Driver, A. (2019). *National Biodiversity Assessment 2018: The status of South Africa's ecosystems and biodiversity* [Synthesis Report]. South African National Biodiversity Institute, an entity of the Department of Environment, Forestry and Fisheries.

Sliwa, C., Migus, S., McEnnulty, F., & Hayes, K. R. (2008). Marine Bioinvasions in Australia. In G. Rilov & J. A. Crooks (Eds.), *Biological Invasions in Marine Ecosystems* (Vol. 204, pp. 425–437). Springer, Berlin. https://doi.org/10.1007/978-3-540-79236-9_25

Sloan, S., & Sayer, J. A. (2015). Forest Resources Assessment of 2015 shows positive global trends but forest loss and degradation persist in poor tropical countries. *Forest Ecology and Management, 352*, 134–145. https://doi.org/10.1016/j.foreco.2015.06.013

Smelansky, I. E., & Tishkov, A. A. (2012). The Steppe Biome in Russia: Ecosystem Services, Conservation Status, and Actual Challenges. In M. J. A. Werger & M. A. van Staalduinen (Eds.), *Eurasian Steppes. Ecological Problems and Livelihoods in a Changing World* (Vol. 6, pp. 45–101). Springer Netherlands. https://doi.org/10.1007/978-94-007-3886-7_2

Smith, C. R., Grange, L. J., Honig, D. L., Naudts, L., Huber, B., Guidi, L., & Domack, E. (2012). A large population of king crabs in Palmer Deep on the west Antarctic Peninsula shelf and potential invasive impacts. *Proceedings of the Royal Society B: Biological Sciences, 279*(1730), 1017–1026. https://doi.org/10.1098/rspb.2011.1496

Smith, J. E. (2011). Algae. In D. Simberloff & M. Rejmánek (Eds.), *Encyclopedia of biological invasions* (Vol. 3). University of California Press. https://doi.org/10.1525/9780520948433

Smith, R. M., Baker, R. H. A., Collins, D. W., Korycinska, A., Malumphy, C. P., Ostojá-Starzewski, J. C., Prior, T., Pye, D., & Reid, S. (2018). Recent trends in non-native, invertebrate, plant pest establishments in Great Britain, accounting for time lags in reporting. *Agricultural and Forest Entomology, 20*(4), 496–504. https://doi.org/10.1111/afe.12282

Sohrabi, S., Pergl, J., Pyšek, P., Foxcroft, L. C., & Gherekhloo, J. (2021). Quantifying the potential impact of alien plants of Iran using the Generic Impact Scoring System (GISS) and Environmental Impact Classification for Alien Taxa (EICAT). *Biological Invasions, 23*(8), 2435–2449. https://doi.org/10.1007/s10530-021-02515-6

Solovjova, N. V. (2019). Ecological risk modelling in developing resources of ecosystems characterized by varying vulnerability levels. *Ecological Modelling, 406*, 60–72. https://doi.org/10.1016/j.ecolmodel.2019.05.015

Soorae, P. S., Al Quarqaz, M., & Gardner, A. S. (2010). An overview and checklist of the native and alien herpetofauna of the United Arab Emirates. *Herpetological Conservation and Biology, 5*(3), 529–536. https://www.herpconbio.org/Volume_5/Issue_3/Soorae_etal_2010.pdf

Soto, D., Jara, F., & Moreno, C. (2001). Escaped salmon in the inner seas, southern Chile: Facing ecological and social conflicts. *Ecological Applications, 11*(6), 1750–1762. https://doi.org/10.1890/1051-0761(2001)011[1750:ESITIS]2.0.CO;2

Soubeyran, Y., Meyer, J.-Y., Lebouvier, M., De Thoisy, B., Lavergne, C., Urtizberea, F., & Kirchner, F. (2015). Dealing with invasive alien species in the French overseas territories: Results and benefits of a 7-year Initiative. *Biological Invasions, 17*(2), 545–554. https://doi.org/10.1007/s10530-014-0766-2

Sousa, A., Laranjeiro, F., Takahashi, S., Tanabe, S., & Barroso, C. M. (2009). Imposex and organotin prevalence in a European post-legislative scenario: Temporal trends from 2003 to 2008. *Chemosphere, 77*(4), 566–573. https://doi.org/10.1016/j.chemosphere.2009.06.049

Sousa, R., Gutiérrez, J. L., & Aldridge, D. C. (2009). Non-indigenous invasive bivalves as ecosystem engineers. *Biological Invasions, 11*(10), 2367–2385. https://doi.org/10.1007/s10530-009-9422-7

Sousa, R., Novais, A., Costa, R., & Strayer, D. L. (2014). Invasive bivalves in fresh waters: Impacts from individuals to ecosystems and possible control strategies. *Hydrobiologia, 735*(1), 233–251. https://doi.org/10.1007/s10750-012-1409-1

South, J., Madzivanzira, T. C., Tshali, N., Measey, J., & Weyl, O. L. F. (2020). In a pinch: Mechanisms behind potential biotic resistance toward two invasive crayfish by native African freshwater crabs. *Frontiers in Ecology and Evolution, 8*, 72. https://doi.org/10.3389/fevo.2020.00072

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdaña, Z. A., Finlayson, M., Halpern, B. S., Jorge, M. A., Lombana, A., Lourie, S. A., Martin, K. D., McManus, E., Molnar, J., Recchia, C. A., & Robertson, J. (2007). Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. *BioScience, 57*(7), 573–583. https://doi.org/10.1641/B570707

Spatz, D. R., Zilliacus, K. M., Holmes, N. D., Butchart, S. H. M., Genovesi, P., Ceballos, G., Tershy, B. R., & Croll, D. A. (2017). Globally threatened vertebrates on islands with invasive species. *Science Advances, 3*(10), e1603080. https://doi.org/10.1126/sciadv.1603080

Spear, D., & Chown, S. L. (2009). The extent and impacts of ungulate translocations: South Africa in a global context. *Biological Conservation, 142*(2), 353–363. https://doi.org/10.1016/j.biocon.2008.10.031

Spear, M. J., Walsh, J. R., Ricciardi, A., & Zanden, M. J. V. (2021). The invasion ecology of sleeper populations: Prevalence, persistence, and abrupt shifts. *BioScience, 71*(4), 357–369. https://doi.org/10.1093/biosci/biaa168

St Clair, J. J. H. (2011). The impacts of invasive rodents on island invertebrates. *Biological Conservation, 144*(1), 68–81. https://doi.org/10.1016/j.biocon.2010.10.006

Stalter, R., Kincaid, D. T., & Lamont, E. E. (1992). Alien plants at Orient Beach State Park, New York, and a comparison with five coastal sites. *Proceedings, 46th Annual Meeting of the Northeasten Weed Science Society*, 106–110. https://www.cabdirect.org/cabdirect/abstract/19922320105

Stevens, P., & van Elsas, J. D. (2010). A putative genomic island, PGI-1, in *Ralstonia solanacearum* biovar 2 revealed by subtractive hybridization. *Antonie van Leeuwenhoek, 98*(3), 359–377. https://doi.org/10.1007/s10482-010-9450-4

Steyn, C., Greve, M., Robertson, M. P., Kalwij, J. M., & le Roux, P. C. (2017). Alien plant species that invade high elevations are generalists: Support for the directional ecological filtering hypothesis. *Journal of Vegetation Science, 28*(2), 337–346. https://doi.org/10.1111/jvs.12477

Stoscheck, L. M., Sherman, R. E., Suarez, E. R., & Fahey, T. J. (2012). Exotic earthworm distributions did not expand over a decade in a hardwood forest in New York state. *Applied Soil Ecology, 62*, 124–130. https://doi.org/10.1016/j.apsoil.2012.07.002

Strayer, D. L. (2010). Alien species in fresh waters: Ecological effects, interactions with other stressors, and prospects for the future. *Freshwater Biology, 55*, 152–174. https://doi.org/10.1111/j.1365-2427.2009.02380.x

Strayer, D. L., D'Antonio, C. M., Essl, F., Fowler, M. S., Geist, J., Hilt, S., Jarić, I., Jöhnk, K., Jones, C. G., Lambin, X., Latzka, A. W., Pergl, J., Pyšek, P., Robertson, P., von Schmalensee, M., Stefansson, R. A., Wright, J., & Jeschke, J. M. (2017). Boom-bust dynamics in biological invasions: Towards an improved application of the concept. *Ecology Letters, 20*(10), 1337–1350. https://doi.org/10.1111/ele.12822

Strayer, D. L., Solomon, C. T., Findlay, S. E. G., & Rosi, E. J. (2019). Long-term research reveals multiple relationships between the abundance and impacts of a non-native species. *Limnology and Oceanography, 64*(S1), S105–S117. https://doi.org/10.1002/lno.11029

Stringham, O. C., & Lockwood, J. L. (2018). Pet problems: Biological and economic factors that influence the release of alien reptiles and amphibians by pet owners. *Journal of Applied Ecology, 55*(6), 2632–2640. https://doi.org/10.1111/1365-2664.13237

Strong, D. R., & Ayres, D. R. (2013). Ecological and evolutionary misadventures of *Spartina. Annual Review of Ecology, Evolution, and Systematics, 44*(1), 389–410. https://doi.org/10.1146/annurev-ecolsys-110512-135803

Stuart, S. N., Hoffmann, M., Chanson, J. S., Cox, N. A., Berridge, R. J., Ramani, P., & Young, B. E. (Eds.). (2008). *Threatened Amphibian Species of the World.* Lynx Edicions, IUCN, Conservation International.

Suarez, A. V., Holway, D. A., & Ward, P. S. (2005). The role of opportunity in the unintentional introduction of nonnative ants. *Proceedings of the National Academy of Sciences of the United States of America, 102*(47), 17032–17035. https://doi.org/10.1073/pnas.0506119102

Suarez, A. V., & Tsutsui, N. D. (2008). The evolutionary consequences of biological invasions. *Molecular Ecology, 17*(1), 351–360. https://doi.org/10.1111/j.1365-294X.2007.03456.x

Suárez, E. R., Fahey, T. J., Groffman, P. M., Yavitt, J. B., & Bohlen, P. J. (2006). Spatial and temporal dynamics of exotic earthworm communities along invasion fronts in a temperate hardwood forest in South-Central New York (USA). *Biological Invasions, 8*(4), 553–564. https://doi.org/10.1007/s10530-005-1196-y

Suda, J., Meyerson, L. A., Leitch, I. J., & Pyšek, P. (2015). The hidden side of plant invasions: The role of genome size. *New Phytologist, 205*(3), 994–1007. https://doi.org/10.1111/nph.13107

Sugiura, S. (2010). Prey preference and gregarious attacks by the invasive flatworm *Platydemus manokwari. Biological Invasions, 12*(6), 1499–1507. https://doi.org/10.1007/s10530-009-9562-9

Sugiura, S., & Yamaura, Y. (2009). Potential impacts of the invasive flatworm *Platydemus manokwari* on arboreal snails. *Biological Invasions, 11*(3), 737–742. https://doi.org/10.1007/s10530-008-9287-1

Sujanapal, P., & Sankaran, K. V. (2016). *Common plants of Maldives*. Food and Agriculture Organization of the United Nations and Kerala Forest Research Institute. https://www.fao.org/publications/card/en/c/a6684e24-04f6-4f1a-9f74-ff9257430139/

Sukhorukov, A. P. (2011). New invasive alien plant species in the forest-steppe and northern steppe subzones of European Russia: Secondary range patterns, ecology and causes of fragmentary distribution. *Feddes Repertorium, 122*(3–4), 287–304. https://doi.org/10.1002/fedr.201100004

Sukopp, H. (2002). On the Early History of Urban Ecology in Europe. In J. M. Marzluff, E. Shulenberger, W. Endlicher, M. Alberti, G. Bradley, C. Ryan, U. Simon, & C. ZumBrunnen (Eds.), *Urban Ecology* (Vol. 74, pp. 79–97). Springer. https://doi.org/10.1007/978-0-387-73412-5_6

Sullivan, J. T., Eardly, B. D., van Berkum, P., & Ronson, C. W. (1996). Four unnamed species of nonsymbiotic rhizobia isolated from the rhizosphere of *Lotus corniculatus. Applied and Environmental Microbiology, 62*(8), 2818–2825. https://doi.org/10.1128/AEM.62.8.2818-2825.1996

Sullivan, J. T., Patrick, H. N., Lowther, W. L., Scott, D. B., & Ronson, C. W. (1995). Nodulating strains of *Rhizobium loti* arise through chromosomal symbiotic gene transfer in the environment. *Proceedings of the National Academy of Sciences of the United States of America, 92*(19), 8985–8989. https://doi.org/10.1073/pnas.92.19.8985

Sundet, J. H., & Hjelset, A. M. (2011). Seasonal Depth Distribution of the Red King Crab *(Paralithodes camtschaticus)* in Varangerfjorden, Northern Norway. In G. H. Kruse, G. L. Eckert, R. J. Foy, R. N. Lipcius, B. Sainte-Marie, D. L. Stram, & D. Woodby (Eds.), *Biology and Management of Exploited Crab Populations under Climate Change* (pp. 403–412). Alaska Sea Grant, University of Alaska Fairbanks. https://doi.org/10.4027/bmecpcc.2010.20

Taheri, A., Elmahroussi, M., Reyes-López, J.-L., Bennas, N., & Brito, J. C. (2021). Ants invading deserts: Non-native species in arid Moroccan oases. *Journal of Arid Environments, 184*, 104122. https://doi.org/10.1016/j.jaridenv.2020.104122

Taheri, S., Pelosi, C., & Dupont, L. (2018). Harmful or useful? A case study of the exotic peregrine earthworm morphospecies *Pontoscolex corethrurus. Soil Biology and Biochemistry, 116*, 277–289. https://doi.org/10.1016/j.soilbio.2017.10.030

Talamas, E. J., Herlihy, M. V., Dieckhoff, C., Hoelmer, K. A., Buffington, M., Bon, M.-C., & Weber, D. C. (2015). *Trissolcus japonicus* (Ashmead) (Hymenoptera, Scelionidae) emerges in North America. *Journal of Hymenoptera Research, 43*, 119–128. https://doi.org/10.3897/JHR.43.4661

Tan, D. T., Thu, P. Q., & Dell, B. (2012). Invasive plant species in National Parks of Vietnam. *Forests, 3*(4), 997–1016. https://doi.org/10.3390/f3040997

Tan, H. H., LIM, K. K. P., Liew, J. H., Low, B. W., Lim, B. H. R., Kwik, J. T. B., & Yeo, D. C. J. (2020). The non-native freshwater fishes of Singapore: An annotated compilation. *Raffles Bulletin of Zoology, 68*, 150–195. https://doi.org/10.26107/RBZ-2020-0016

Tanaka, T., & Sato, T. (2016). Contemporary patterns and temporal changes in alien plant species richness along an elevational gradient in central Japan. *Plant Ecology and Evolution, 149*(2), 177–188. https://doi.org/10.5091/plecevo.2016.1197

Tang, Q. S., Fang, J. G., & Liu, H. (2002). Development of mussel aquaculture in China. *Bulletin of the Aquaculture Association of Canada, 102*(3), 66–74. https://www.cabi.org/isc/abstract/20033089587

Tassin, J., & Laizé, A. (2015). Facteurs biologiques et structuraux de l'invasion de la liane *Merremia peltata* dans les habitats forestiers de mayotte. *Revue d'Ecologie (Terre et Vie), 70*(suppl. 12 Espèces invasives), 151–161. https://agritrop.cirad.fr/581483/

Tassin, J., Rangan, H., & Kull, C. A. (2012). Hybrid improved tree fallows: Harnessing invasive woody legumes for agroforestry. *Agroforestry Systems, 84*(3), 417–428. https://doi.org/10.1007/s10457-012-9493-9

Taylor, C. M., & Hastings, A. (2005). Allee effects in biological invasions. *Ecology Letters, 8*(8), 895–908. https://doi.org/10.1111/j.1461-0248.2005.00787.x

Taylor, S., & Kumar, L. (2016). Will climate change impact the potential distribution of a native vine *(Merremia peltata)* which is behaving invasively in the Pacific region? *Ecology and Evolution, 6*(3), 742–754. https://doi.org/10.1002/ece3.1915

TDWG. (2021). *Biodiversity Information Standards (TDWG)*. Biodiversity Information Standards (TDWG). https://www.tdwg.org/

te Beest, M., Le Roux, J. J., Richardson, D. M., Brysting, A. K., Suda, J., Kubesova, M., & Pyšek, P. (2011). The more the better? The role of polyploidy in facilitating plant invasions. *Annals of Botany, 109*(1), 19–45. https://doi.org/10.1093/aob/mcr277

Tedeschi, L., Biancolini, D., Capinha, C., Rondinini, C., & Essl, F. (2022). Invasive alien mammals of European Union concern. *Mammal Review, 52*(2), 252–266. https://doi.org/10.1111/mam.12277

Teixeira, L. M. P., & Creed, J. C. (2020). A decade on: An updated assessment of the status of marine non-indigenous species in Brazil. *Aquatic Invasions, 15*(1), 30–43. https://doi.org/10.3391/ai.2020.15.1.03

Teletchea, F. (2019). Fish domestication in aquaculture: Reassessment and emerging questions. *Cybium, 43*(1), 7–15. https://doi.org/10.26028/cybium/2019-431-001

Teletchea, F., & Beisel, J.-N. (2018). *Alien Fish Species in France with Emphasis on the Recent Invasion of Gobies.* IntechOpen. https://doi.org/10.5772/intechopen.73408

Telford, N. S., Channing, A., & Measey, J. (2019). Origin of invasive populations of the Guttural Toad *(Sclerophrys gutturalis)* on Réunion and Mauritius Islands and in Constantia, South Africa. *Herpetological Conservation and Biology, 14*(2), 380–392. https://www.herpconbio.org/Volume_14/Issue_2/Telford_etal_2019.pdf

Temple, S. A. (1992). Exotic birds: A growing problem with no easy solution. *The Auk, 109*(2), 395–397. https://doi.org/10.2307/4088214

Tershy, B. R., Shen, K.-W., Newton, K. M., Holmes, N. D., & Croll, D. A. (2015). The importance of islands for the protection of biological and linguistic diversity. *BioScience, 65*(6), 592–597. https://doi.org/10.1093/biosci/biv031

Texas Invasive Species Institute. (2021). *Barbary sheep*. http://www.tsusinvasives.org/home/database/ammotragus-lervia

Thakur, M. P., van der Putten, W. H., Cobben, M. M. P., van Kleunen, M., & Geisen, S. (2019). Microbial invasions in terrestrial ecosystems. *Nature Reviews Microbiology, 17*(10), 621–631. https://doi.org/10.1038/s41579-019-0236-z

Thaman, R. R. (2011). The silent invasion of our islands. *Mai Life, 55*(Dec.), 64–65. http://repository.usp.ac.fj/id/eprint/7397

Thaman, R. R., & O'Brien, K. (2011). Caterpillar devastates kanava and undermines resilience to climate change in Tuvalu. *Mai Life, 50*(July), 56–57.

The Ministry of the Environment of Japan. (2014). *Mongoose eradication project in Yambaru, Okinawa: For restoring forest ecosystem and native animals in Yambaru* (Nansei Environmental Laboratory Co., Ltd., Ed.). The Ministry of the Environment, Naha Nature Conservation Office. https://kyushu.env.go.jp/okinawa/files/mongoose_yambaru_E.pdf

The Plant List. (2015). *Home—The Plant List.* The Plant List: A Working List of All Plant Species. http://www.theplantlist.org/

Therriault, T. W., Campbell, M. L., Deidun, A., Galil, B. S., Hewitt, C. L., Inglis, G. J., Ojaveer, H., Park, C., Qiao, B., Ruwa, R. K., & Schwindt, E. (2021). Invasive species. In *The Second World Ocean Assessment* (pp. 343–362). United Nations. https://researchrepository.murdoch.edu.au/id/eprint/64636/

Therriault, T. W., Nelson, J. C., Carlton, J. T., Liggan, L., Otani, M., Kawai, H., Scriven, D., Ruiz, G. M., & Murray, C. C. (2018). The invasion risk of species associated with Japanese Tsunami Marine Debris in Pacific North America and Hawaii. *Marine Pollution Bulletin, 132*, 82–89. https://doi.org/10.1016/j.marpolbul.2017.12.063

Thieltges, D. W. (2005a). Benefit from an invader: American slipper limpet *Crepidula fornicata* reduces star fish predation on basibiont European mussels. *Hydrobiologia, 541*(1), 241–244. https://doi.org/10.1007/s10750-004-4671-z

Thieltges, D. W. (2005b). Impact of an invader: Epizootic American slipper limpet *Crepidula fornicata* reduces survival and growth in European mussels. *Marine Ecology Progress Series, 286*, 13–19. https://doi.org/10.3354/meps286013

Thines, M. (2011). Recent outbreaks of downy mildew on grape ivy (*Parthenocissus tricuspidata*, Vitaceae) in Germany are caused by a new species of *Plasmopara. Mycological Progress, 10*(4), 415–422. https://doi.org/10.1007/s11557-010-0712-3

Thines, M. (2014). Phylogeny and evolution of plant pathogenic oomycetes—A global overview. *European Journal of Plant Pathology, 138*(3), 431–447. https://doi.org/10.1007/s10658-013-0366-5

Thines, M., Buaya, A., Ali, T., & Brand, T. (2020). *Peronospora aquilegiicola* made its way to Germany: The start of a new pandemic? *Mycological Progress, 19*(8), 791–798. https://doi.org/10.1007/s11557-020-01596-2

Thines, M., & Choi, Y.-J. (2016). Evolution, Diversity, and Taxonomy of the Peronosporaceae, with Focus on the Genus *Peronospora. Phytopathology, 106*(1), 6–18. https://doi.org/10.1094/PHYTO-05-15-0127-RVW

Thomsen, C. N., & Hart, M. M. (2018). Using invasion theory to predict the fate of arbuscular mycorrhizal fungal inoculants. *Biological Invasions, 20*(10), 2695–2706. https://doi.org/10.1007/s10530-018-1746-8

Thorn, J. P. R. (2019). Adaptation "from below" to changes in species distribution, habitat and climate in agro-ecosystems in the Terai Plains of Nepal. *Ambio, 48*, 1482–1497. https://doi.org/10.1007/s13280-019-01202-0

Thorstad, E. B., Fleming, I. A., McGinnity, P., Soto, D., Wennevik, V., & Whoriskey, F. (2008). *Incidence and impacts of escaped farmed Atlantic salmon* Salmo salar *in nature* (NINA Special Report 36; Technical Working Group on Escapes of the Salmon Aquaculture Dialogue, p. 110). Norwegian Institute for Nature Research (NINA).

Tilahun, M., Birner, R., & Ilukor, J. (2017). Household-level preferences for mitigation of *Prosopis juliflora* invasion in the Afar region of Ethiopia: A contingent valuation. *Journal of Environmental Planning and Management, 60*(2), 282–308. https://doi.org/10.1080/09640568.2016.1152955

Tingley, R., García-Díaz, P., Arantes, C. R. R., & Cassey, P. (2018). Integrating transport pressure data and species distribution models to estimate invasion risk for alien stowaways. *Ecography, 41*(4), 635–646. https://doi.org/10.1111/ecog.02841

Tingley, R., Greenlees, M., Oertel, S., van Rooyen, A. R., & Weeks, A. R. (2019). Environmental DNA sampling as a surveillance tool for cane toad *Rhinella marina* introductions on offshore islands. *Biological Invasions, 21*(1), 1–6. https://doi.org/10.1007/s10530-018-1810-4

Tingley, R., Mahoney, P. J., Durso, A. M., Tallian, A. G., Morán-Ordóñez, A., & Beard, K. H. (2016). Threatened and invasive reptiles are not two sides of the same coin. *Global Ecology and Biogeography, 25*(9), 1050–1060. https://doi.org/10.1111/geb.12462

Tingley, R., Phillips, B. L., & Shine, R. (2011). Establishment success of introduced amphibians increases in the presence of congeneric species. *The American Naturalist, 177*(3), 382–388. https://doi.org/10.1086/658342

Tingley, R., Romagosa, C. M., Kraus, F., Bickford, D., Phillips, B. L., & Shine, R. (2010). The frog filter: Amphibian introduction bias driven by taxonomy, body size and biogeography. *Global Ecology and Biogeography, 19*(4), 496–503. https://doi.org/10.1111/j.1466-8238.2010.00530.x

Tiwari, S., Siwakoti, M., Adhikari, B., & Subedi, K. (2005). *An Inventory and Assessment of Invasive Alien Plant Species of Nepal.* IUCN – The World Conservation Union. https://portals.iucn.org/library/sites/library/files/documents/2005-054.pdf

Tobin, P. C., Bai, B. B., Eggen, D. A., & Leonard, D. S. (2012). The ecology, geopolitics, and economics of managing *Lymantria dispar* in the United States. *International Journal of Pest Management, 58*(3), 195–210. https://doi.org/10.1080/09670874.2011.647836

Tognetti, P. M., & Chaneton, E. J. (2012). Invasive exotic grasses and seed arrival limit native species establishment in an old-field grassland succession. *Biological Invasions, 14*(12), 2531–2544. https://doi.org/10.1007/s10530-012-0249-2

Tolvanen, A., & Kangas, K. (2016). Tourism, biodiversity and protected areas—Review from northern Fennoscandia. *Journal of Environmental Management, 169*, 58–66. https://doi.org/10.1016/j.jenvman.2015.12.011

Toomes, A., García-Díaz, P., Wittmann, T. A., Virtue, J., & Cassey, P. (2020). New aliens in Australia: 18 years of vertebrate interceptions. *Wildlife Research, 47*(1), 55–67. https://doi.org/10.1071/WR18185

Toral-Granda, M. V., Causton, C. E., Jäger, H., Trueman, M., Izurieta, J. C., Araujo, E., Cruz, M., Zander, K. K., Izurieta, A., & Garnett, S. T. (2017). Alien species pathways to the Galapagos Islands, Ecuador. *PLoS ONE, 12*(9), e0184379. https://doi.org/10.1371/journal.pone.0184379

Torchyk, O., & Jeschke, J. M. (2018). Phenotypic plasticity hypothesis. In J. M. Jeschke & T. Heger (Eds.), *Invasion biology: Hypotheses and evidence* (pp. 133–139). CABI International. https://doi.org/10.1079/9781780647647.0133

Totland, Ø., Nyeko, P., Bjerknes, A.-L., Hegland, S. J., & Nielsen, A. (2005). Does forest gap size affects population size, plant size, reproductive success and pollinator visitation in *Lantana camara*, a tropical invasive shrub? *Forest Ecology and Management, 215*(1–3), 329–338. https://doi.org/10.1016/j.foreco.2005.05.023

Tousignant, M.-Ê., Pellerin, S., & Brisson, J. (2010). The relative impact of human disturbances on the vegetation of a large wetland complex. *Wetlands, 30*(2), 333–344. https://doi.org/10.1007/s13157-010-0019-9

Toussaint, A., Beauchard, O., Oberdorff, T., Brosse, S., & Villéger, S. (2016). Worldwide freshwater fish homogenization is driven by a few widespread non-native species. *Biological Invasions, 18*(5), 1295–1304. https://doi.org/10.1007/s10530-016-1067-8

Towns, D. R., Atkinson, I. A. E., & Daugherty, C. H. (2006). Have the harmful effects of introduced rats on islands been exaggerated? *Biological Invasions, 8*(4), 863–891. https://doi.org/10.1007/s10530-005-0421-z

Tretyakova, A. S., Veselkin, D. V., Senator, S. A., & Golovanov, Ya. M. (2018). Factors of richness of urban floras in the Ural–Volga region. *Russian Journal of Ecology, 49*(3), 201–208. https://doi.org/10.1134/S1067413618030098

Troudet, J., Grandcolas, P., Blin, A., Vignes-Lebbe, R., & Legendre, F. (2017). Taxonomic bias in biodiversity data and societal preferences. *Scientific Reports, 7*(1), 9132. https://doi.org/10.1038/s41598-017-09084-6

Trueman, M., Atkinson, R., Guézou, A., & Wurm, P. (2010). Residence time and human-mediated propagule pressure at work in the alien flora of Galapagos. *Biological Invasions, 12*(12), 3949–3960. https://doi.org/10.1007/s10530-010-9822-8

Tsatsia, H., & Jackson, G. (2022). *Giant African snail (050)*. Pacific Pests, Pathogens, Weed & Pesticides – Online Edition. https://apps.lucidcentral.org/pppw_v11/text/web_full/entities/giant_african_snail_050.htm

Tsiamis, K., Azzurro, E., Bariche, M., Çinar, M. E., Crocetta, F., De Clerck, O., Galil, B. S., Gómez, F., Hoffman, R., Jensen, K. R., Kamburska, L., Langeneck, J., Langer, M. R., Levitt-Barmats, Y., Lezzi, M., Marchini, A., Occhipinti-Ambrogi, A., Ojaveer, H., Piraino, S., … Cardoso, A. C. (2020). Prioritizing marine invasive alien species in the European Union through horizon scanning. *Aquatic Conservation: Marine and Freshwater Ecosystems, 30*(4), 794–845. https://doi.org/10.1002/aqc.3267

Tsiamis, K., Palialexis, A., Connor, D., Antoniadis, S., Bartilotti, C., Bartolo, A., Berggreen, U. C., Boschetti, S., Buschbaum, C., Canning-Clode, J., Carbonell, A., Castriota, L., Corbeau, C., Costa, A., Cvitković, I., Despalatović, M., Dragičević, B., Dulčić, J., Fortič, A., … Žuljević, A. (2021). *Marine Strategy Framework Directive—Descriptor 2, Non-Indigenous Species, Delivering solid recommendations for setting threshold values for non-indigenous species pressure on European seas.* Centro Oceanográfico de Baleares. https://doi.org/10.2760/035071

Tsiamis, K., Palialexis, A., Stefanova, K., Gladan, Ž. N., Skejić, S., Despalatović, M., Cvitković, I., Dragičević, B., Dulčić, J., Vidjak, O., Bojanić, N., Žuljević, A., Aplikioti, M., Argyrou, M., Josephides, M., Michailidis, N., Jakobsen, H. H., Staehr, P. A., Ojaveer, H., … Cardoso, A. C. (2019). Non-indigenous species refined national baseline inventories: A synthesis in the context of the European Union's Marine Strategy Framework Directive. *Marine Pollution Bulletin, 145*, 429–435. https://doi.org/10.1016/j.marpolbul.2019.06.012

Tsiamis, K., Zenetos, A., Deriu, I., Gervasini, E., & Cardoso, A. C. (2018). The native distribution range of the European marine non-indigenous species. *Aquatic Invasions, 13*(2), 187–198. https://doi.org/10.3391/ai.2018.13.2.01

Tsutsui, N. D., Suarez, A. V., Holway, D. A., & Case, T. J. (2000). Reduced genetic variation and the success of an invasive species. *Proceedings of the National Academy of Sciences, 97*(11), 5948–5953. https://doi.org/10.1073/pnas.100110397

Tumushabe, G., & Mugyenyi, O. (2017). *Forest Governance Training manual.* Advocates Coalition for Development and Environment (ACODE). https://www.acode-u.org/uploadedFiles/GovernanceManual.pdf

Turbelin, A. J., Malamud, B. D., & Francis, R. A. (2017). Mapping the global state of invasive alien species: Patterns of invasion and policy responses. *Global Ecology and Biogeography, 26*(1), 78–92. https://doi.org/10.1111/geb.12517

Turchini, G. M., & De Silva, S. S. (2008). Bio-economical and ethical impacts of alien finfish culture in European inland waters. *Aquaculture International, 16*(3), 243–272. https://doi.org/10.1007/s10499-007-9141-y

Turner, S. C., Esler, K. J., & Kalwij, J. M. (2021). Road verges facilitate exotic species' expansion into undisturbed natural montane grasslands. *Applied Vegetation Science, 24*(4), e12615. https://doi.org/10.1111/avsc.12615

Ugarte, E., Fuentes, N., & Klotz, S. (2010). European plant in southern South America, unwanted visitors? In J. Settele (Ed.), *Atlas of Biodiversity Risk* (pp. 148–150). Pensoft Publishers. https://www.nhbs.com/atlas-of-biodiversity-risk-book

Ulman, A., Ferrario, J., Forcada, A., Seebens, H., Arvanitidis, C., Occhipinti-Ambrogi, A., & Marchini, A. (2019). Alien species spreading via biofouling on recreational vessels in the Mediterranean Sea. *Journal of Applied Ecology, 56*(12), 2620–2629. https://doi.org/10.1111/1365-2664.13502

UNEP-WCMC, UNEP, & IUCN. (2021). *Protected Planet Report 2020*. UNEP World Conservation Monitoring Centre, International Union for Conservation of Nature. https://doi.org/10.34892/JG6T-XN70

Urbina, J. C., & Benavides, J. C. (2015). Simulated small scale disturbances increase decomposition rates and facilitates invasive species encroachment in a high elevation tropical Andean peatland. *Biotropica, 47*(2), 143–151. https://doi.org/10.1111/btp.12191

Urgamal, M., Oyuntsetseg, B., Nyambayar, D., & Dulamsuren, C. (2014). *Conspectus of the vascular plants of Mongolia* (Ch. Sanchir & Ts. Jamsran, Eds.). Admon Press.

USDA, NRCS. (2021, March 9). *USDA Plants Database* [United States Department of Agriculture, Natural Resources Conservation Service]. The PLANTS Database. https://plants.usda.gov/home

USGS. (2021). *Nonindigenous Aquatic Species (NAS)* [United States Geological Survey]. https://nas.er.usgs.gov/

Usher, M. B. (Ed.). (1986). *Wildlife conservation evaluation*. Springer Netherlands. https://doi.org/10.1007/978-94-009-4091-8

Usher, M. B. (1988). Biological invasions of nature reserves: A search for generalisations. *Biological Conservation, 44*(1–2), 119–135. https://doi.org/10.1016/0006-3207(88)90007-9

Usher, M. B. (2005). Principles of conserving the Arctic's biodiversity. In *Arctic Climate Impact Assessment* (pp. 539–596). Cambridge University Press.

Vakhlamova, T., Rusterholz, H.-P., Kanibolotskaya, Y., & Baur, B. (2016). Effects of road type and urbanization on the diversity and abundance of alien species in roadside verges in Western Siberia. *Plant Ecology, 217*(3), 241–252. https://doi.org/10.1007/s11258-016-0565-1

Valenta, V., Moser, D., Kapeller, S., & Essl, F. (2017). A new forest pest in Europe: A review of Emerald ash borer *(Agrilus planipennis)* invasion. *Journal of Applied Entomology, 141*(7), 507–526. https://doi.org/10.1111/jen.12369

Van der Burg, W. J., de Freitas, J., Debrot, A. O., & Lotz, L. A. P. (2012). *Naturalised and invasive alien plant species in the Caribbean Netherlands: Status, distribution, threats, priorities and recommendations: Report of a joint Imares/Carmabi/PRI project financed by the Dutch Ministry of Economic Affairs, Agriculture & Innovation* (Plant Research International Report 437 C185/11). Plant Research International, Business Unit Agrosystems Research. https://library.wur.nl/WebQuery/wurpubs/fulltext/198282

Van Keer, K. (2010). An update on the verified reports of imported spiders (Araneae) from Belgium. *Nieuwsbrief van de Belgische Arachnologische Vereniging, 25*(3), 210–214. https://araneae.nmbe.ch/pdfs/45031_Van_Keer_2011_NieuwsbrBelgArachnolVer_25_3_2010_p210-214_Araneae_imported_Belgium.pdf

van Kleunen, M., Dawson, W., Essl, F., Pergl, J., Winter, M., Weber, E., Kreft, H., Weigelt, P., Kartesz, J., Nishino, M., Antonova, L. A., Barcelona, J. F., Cabezas, F. J., Cárdenas, D., Cárdenas-Toro, J., Castaño, N., Chacón, E., Chatelain, C., Ebel, A. L., … Pyšek, P. (2015). Global exchange and accumulation of non-native plants. *Nature, 525*(7567), 100–103. https://doi.org/10.1038/nature14910

van Kleunen, M., Essl, F., Pergl, J., Brundu, G., Carboni, M., Dullinger, S., Early, R., González-Moreno, P., Groom, Q. J., Hulme, P. E., Kueffer, C., Kühn, I., Máguas, C., Maurel, N., Novoa, A., Parepa, M., Pyšek, P., Seebens, H., Tanner, R., … Dehnen-Schmutz, K. (2018). The changing role of ornamental horticulture in alien plant invasions. *Biological Reviews, 93*(3), 1421–1437. https://doi.org/10.1111/brv.12402

van Kleunen, M., Pyšek, P., Dawson, W., Essl, F., Kreft, H., Pergl, J., Weigelt, P., Stein, A., Dullinger, S., König, C., Lenzner, B., Maurel, N., Moser, D., Seebens, H., Kartesz, J., Nishino, M., Aleksanyan, A., Ansong, M., Antonova, L. A., … Winter, M. (2019). The Global Naturalized Alien Flora (GloNAF) database. *Ecology, 100*(1), e02542. https://doi.org/10.1002/ecy.2542

Van Poucke, K., Haegeman, A., Goedefroit, T., Focquet, F., Leus, L., Jung, M. H., Nave, C., Redondo, M. A., Husson, C., Kostov, K., Lyubenova, A., Christova, P., Chandelier, A., Slavov, S., de Cock, A., Bonants, P., Werres, S., Palau, J. O., Marçais, B., … Heungens, K. (2021). Unravelling hybridization in *Phytophthora* using phylogenomics and genome size estimation. *IMA Fungus, 12*(1), 16. https://doi.org/10.1186/s43008-021-00068-w

van Wesenbeeck, B. K., van Mourik, T., Duivenvoorden, J. F., & Cleef, A. M. (2003). Strong effects of a plantation with *Pinus patula* on Andean subpáramo vegetation: A case study from Colombia. *Biological Conservation, 114*(2), 207–218. https://doi.org/10.1016/S0006-3207(03)00025-9

van Wilgen, B. W. (2012). Evidence, perceptions, and trade-offs associated with invasive alien plant control in the Table Mountain National Park, South Africa. *Ecology and Society, 17*(2), 23. https://doi.org/10.5751/ES-04590-170223

van Wilgen, B. W. (2018). The management of invasive alien plants in South Africa: Strategy, progress and challenges. *Outlooks on Pest Management, 29*(1), 13–17. https://doi.org/10.1564/v29_feb_04

van Wilgen, B. W., Fill, J. M., Baard, J., Cheney, C., Forsyth, A. T., & Kraaij, T. (2016). Historical costs and projected future scenarios for the management of invasive alien plants in protected areas in the Cape Floristic Region. *Biological Conservation, 200*, 168–177. https://doi.org/10.1016/j.biocon.2016.06.008

van Wilgen, B. W., Measey, J., Richardson, D. M., Wilson, J. R., & Zengeya, T. A. (Eds.). (2020). *Biological Invasions in South Africa.* Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3

van Wilgen, B. W., & Wilson, J. R. (Eds.). (2018). *The status of biological invasions and their management in South Africa in 2017* (p. 398). South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology.

van Wilgen, N. J., Gillespie, M. S., Richardson, D. M., & Measey, J. (2018). A taxonomically and geographically constrained information base limits non-native reptile and amphibian risk assessment: A systematic review. *PeerJ, 6*, e5850. https://doi.org/10.7717/peerj.5850

Van Wilgen, N. J., & Richardson, D. M. (2012). The roles of climate, phylogenetic relatedness, introduction effort, and reproductive traits in the establishment of non-native reptiles and amphibians. *Conservation Biology, 26*(2), 267–277. https://doi.org/10.1111/j.1523-1739.2011.01804.x.

Van Wilgen, N. J., Wilson, J. R. U., Elith, J., Wintle, B. A., & Richardson, D. M. (2010). Alien invaders and reptile traders: What drives the live animal trade in South Africa? *Animal Conservation, 13*(s1), 24–32. https://doi.org/10.1111/j.1469-1795.2009.00298.x

Vancutsem, C., Achard, F., Pekel, J.-F., Vieilledent, G., Carboni, S., Simonetti, D., Gallego, J., Aragão, L. E. O. C., & Nasi, R. (2021). Long-term (1990–2019) monitoring of forest cover changes in the humid tropics. *Science Advances, 7*(10), eabe1603. https://doi.org/10.1126/sciadv.abe1603

Vanderploeg, H. A., Nalepa, T. F., Jude, D. J., Mills, E. L., Holeck, K. T., Liebig, J. R., Grigorovich, I. A., & Ojaveer, H. (2002). Dispersal and emerging ecological impacts of Ponto-Caspian species in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 59*(7), 1209–1228. https://doi.org/10.1139/f02-087

Vaz-Pinto, F., Rodil, I. F., Mineur, F., Olabarria, C., & Arenas, F. (2014). Understanding biological invasions by seaweeds. In L. Pereira & J. M. Neto (Eds.), *Marine Algae. Biodiversity, Taxonomy, Environmental Assessment and Biotechnology.* (0 ed., pp. 140–177). CRC Press. https://doi.org/10.1201/b17540-5

Veitch, C. R., Clout, M. N., Martin, A. R., Russell, J. C., & West, C. J. (Eds.). (2019). *Island invasives: Scaling up to meet the challenge. Proceedings of the international conference on island invasives 2017: Vol. Occasional Paper SSC no. 62.* IUCN, International Union for Conservation of Nature Species Survival Commission (SSC). https://doi.org/10.2305/IUCN.CH.2019.SSC-OP.62.en

Velásquez, A. C., Castroverde, C. D. M., & He, S. Y. (2018). Plant–Pathogen Warfare under Changing Climate Conditions. *Current Biology, 28*(10), R619–R634. https://doi.org/10.1016/j.cub.2018.03.054

Vellend, M., Harmon, L. J., Lockwood, Julie. L., Mayfield, M. M., Hughes, A. R., Wares, J. P., & Sax, D. F. (2007). Effects of exotic species on evolutionary diversification. *Trends in Ecology & Evolution, 22*(9), 481–488. https://doi.org/10.1016/j.tree.2007.02.017

Vellinga, E. C., Wolfe, B. E., & Pringle, A. (2009). Global patterns of ectomycorrhizal introductions. *New Phytologist, 181*(4), 960–973. https://doi.org/10.1111/j.1469-8137.2008.02728.x

Vences, M., Brown, J. L., Lathrop, A., Rosa, G. M., Cameron, A., Crottini, A., Dolch, R., Edmonds, D., Freeman, K. L., Glaw, F., Grismer, L. L., Litvinchuk, S., Milne, M., Moore, M., Solofo, J. F., Noël, J., Nguyen, T. Q., Ohler, A., Randrianantoandro, C., … Murphy, R. W. (2017). Tracing a toad invasion: Lack of mitochondrial DNA variation, haplotype origins, and potential distribution of introduced *Duttaphrynus melanostictus* in Madagascar. *Amphibia-Reptilia, 38*(2), 197–207. https://doi.org/10.1163/15685381-00003104

Vergés, A., Comalada, N., Sánchez, N., & Brodie, J. (2013). A reassessment of the foliose Bangiales (Rhodophyta) in the Balearic Islands including the proposed synonymy of *Pyropia olivii* with *Pyropia koreana. Botanica Marina, 56*(3). https://doi.org/10.1515/bot-2013-0003

Verloove, F. (2006). *Catalogue of neophytes in Belgium (1800-2005)* (Vol. 39). Meise, National Botanic Garden of Belgium. https://alienplantsbelgium.myspecies.info/sites/alienplantsbelgium.be/files/tabel_2.pdf

Vermeij, G. J., & Roopnarine, P. D. (2008). The Coming Arctic Invasion. *Science, 321*(5890), 780–781. https://doi.org/10.1126/science.1160852

Vidal-Russell, R., Fernández Cánepa, G., Nuñez, C. I., & Ezcurra, C. (2019). First report of alien *Drosera rotundifolia* in a high conservation value Patagonian peat bog. *Weed Research, 59*(6), 458–466. https://doi.org/10.1111/wre.12383

Vignon, M., & Sasal, P. (2010). Fish introduction and parasites in marine ecosystems: A need for information. *Environmental Biology of Fishes, 87*(1), 1–8. https://doi.org/10.1007/s10641-009-9553-9

Vilà, M., Pino, J., & Font, X. (2007). Regional assessment of plant invasions across different habitat types. *Journal of Vegetation Science, 18*(1), 35–42. https://www.jstor.org/stable/4499196

Villamagna, A. M., & Murphy, B. R. (2010). Ecological and socio-economic impacts of invasive water hyacinth *(Eichhornia crassipes)*: A review. *Freshwater Biology, 55*(2), 282–298. https://doi.org/10.1111/j.1365-2427.2009.02294.x

Villaseñor-Parada, C., Macaya, E. C., Jara-Rojas, L. M., & Neill, P. E. (2013). Variación espacial y temporal en la producción de gametangios en la macroalga exótica *Codium fragile* subsp. *Tomentosoides* (Chlorophyta: Bryopsidales) en el submareal de Caldera, Chile. *Revista de Biología Marina y Oceanografía, 48*(1), 213–218. https://doi.org/10.4067/S0718-19572013000100021

Villaseñor-Parada, C., & Neill, P. E. (2011). Distribución espacial de epifitos en el talo de la macroalga introducida *Codium fragile* subsp. *Tomentosoides* en el submareal de Caldera. *Revista de Biología Marina y Oceanografía, 46*(2), 257–262. https://doi.org/10.4067/S0718-19572011000200015

Villaseñor-Parada, C., Pauchard, A., & Macaya, E. C. (2017). Ecología de invasiones marinas en Chile continental: ¿Qué sabemos y que nos falta por saber? *Revista de biología marina y oceanografía, 52*(1), 01–17. https://doi.org/10.4067/S0718-19572017000100001

Villaseñor-Parada, C., Pauchard, A., Ramírez, M. E., & Macaya, E. C. (2018). Macroalgas exoticas en la costa de Chile: Patrones espaciales y temporales en el proceso de invasión. *Latin American Journal of Aquatic Research, 46*(1), 147–165. https://doi.org/10.3856/vol46-issue1-fulltext-15

Vinogradov, Yu. K., & Kupriyanov, A. N. (Eds.). (2016). *The black book of the flora of Siberia* (p. 420). Academic publishing house

"Geo." http://kuzbs.ru/images/stories/pdf/izdania/chernaya_kniga_flori_sibiri.pdf

Vinogradova, Y., Pergl, J., Essl, F., Hejda, M., van Kleunen, M., Pyšek, P., & REGIONAL CONTRIBUTORS. (2018). Invasive alien plants of Russia: Insights from regional inventories. *Biological Invasions, 20*(8), 1931–1943. https://doi.org/10.1007/s10530-018-1686-3

Visser, V., Wilson, J. R. U., Fish, L., Brown, C., Cook, G. D., & Richardson, D. M. (2016). Much more give than take: South Africa as a major donor but infrequent recipient of invasive non-native grasses. *Global Ecology and Biogeography, 25*(6), 679–692. https://doi.org/10.1111/geb.12445

Vítková, M., Müllerová, J., Sádlo, J., Pergl, J., & Pyšek, P. (2017). Black locust *(Robinia pseudoacacia)* beloved and despised: A story of an invasive tree in Central Europe. *Forest Ecology and Management, 384*, 287–302. https://doi.org/10.1016/j.foreco.2016.10.057

Vitousek, P. M., Loope, L. L., & Stone, C. P. (1987). Introduced species in Hawaii: Biological effects and oppotunities for ecological research. *Trends in Ecology & Evolution, 2*(7), 224–227. https://doi.org/10.1016/0169-5347(87)90026-7

Vitule, J. R. S., Bornatowski, H., Freire, C. A., & Abilhoa, V. (2014). Extralimital introductions of *Salminus brasiliensis* (Cuvier, 1816) (Teleostei, Characidae) for sport fishing purposes: A growing challenge for the conservation of biodiversity in neotropical aquatic ecosystems. *BioInvasions Records, 3*(4), 291–296. https://doi.org/10.3391/bir.2014.3.4.11

Vitule, J. R. S., Freire, C. A., Vazquez, D. P., Nuñez, M. A., & Simberloff, D. (2012). Revisiting the potential conservation value of non-native species. *Conservation Biology: The Journal of the Society for Conservation Biology, 26*(6), 1153–1155. https://doi.org/10.1111/j.1523-1739.2012.01950.x

Vitule, J. R. S., Occhi, T. V. T., Carneiro, L., Daga, V. S., Frehse, F. A., Bezerra, L. A. V., Forneck, S., de Pereira, H. S., Freitas, M. O., Hegel, C. G. Z., Abilhoa, V., Grombone-Guaratini, M. T., Queiroz-Sousa, J., Pivello, V. R., Silva-Matos, D. M., Oliveira, I., Toledo, L. F., Vallejos, M. A. V., Zenni, R. D., … Braga, R. R. (2021). Non-native Species Introductions, Invasions, and Biotic Homogenization in the Atlantic Forest. In M. C. M. Marques & C. E. V. Grelle (Eds.), *The Atlantic Forest* (pp. 269–295). Springer International Publishing. https://doi.org/10.1007/978-3-030-55322-7_13

Vitule, J. R. S., Occhi, T. V. T., Kang, B., Matsuzaki, S.-I., Bezerra, L. A., Daga, V. S., Faria, L., Frehse, F. de A., Walter, F., & Padial, A. A. (2019). Intra-country introductions unraveling global hotspots of alien fish species. *Biodiversity and Conservation, 28*(11), 3037–3043. https://doi.org/10.1007/s10531-019-01815-7

Voglmayr, H., Montes-Borrego, M., & Landa, B. B. (2014). Disentangling *Peronospora* on *Papaver*: Phylogenetics, taxonomy, nomenclature and host range of downy mildew of opium poppy *(Papaver somniferum)* and related species. *PLoS ONE, 9*(5), e96838. https://doi.org/10.1371/journal.pone.0096838

Voight, J. R., Lee, R. W., Reft, A. J., & Bates, A. E. (2012). Scientific gear as a vector for non-native species at deep-sea hydrothermal vents. *Conservation Biology, 26*(5), 938–942. https://doi.org/10.1111/j.1523-1739.2012.01864.x

Waage, J. K., Woodhall, J. W., Bishop, S. J., Smith, J. J., Jones, D. R., & Spence, N. J. (2008). Patterns of plant pest introductions in Europe and Africa. *Agricultural Systems, 99*(1), 1–5. https://doi.org/10.1016/j.agsy.2008.08.001

Waddell, E. H., Banin, L. F., Fleiss, S., Hill, J. K., Hughes, M., Jelling, A., Yeong, K. L., Ola, B. B., Sailim, A. B., Tangah, J., & Chapman, D. S. (2020). Land-use change and propagule pressure promote plant invasions in tropical rainforest remnants. *Landscape Ecology, 35*, 1891–1906. https://doi.org/10.1007/s10980-020-01067-9

Wagner, V., Bragina, T. M., Nowak, A., Smelansky, I. E., & Vanselow, K. A. (2020). Grasslands and Shrublands of Kazakhstan and Middle Asia. In *Encyclopedia of the World's Biomes* (pp. 750–758). Elsevier. https://doi.org/10.1016/B978-0-12-409548-9.12043-3

Wagner, V., Chytrý, M., Jiménez-Alfaro, B., Pergl, J., Hennekens, S., Biurrun, I., Knollová, I., Berg, C., Vassilev, K., Rodwell, J. S., Škvorc, Ž., Jandt, U., Ewald, J., Jansen, F., Tsiripidis, I., Botta-Dukát, Z., Casella, L., Attorre, F., Rašomavičius, V., … Pyšek, P. (2017). Alien plant invasions in European woodlands. *Diversity and Distributions, 23*(9), 969–981. https://doi.org/10.1111/ddi.12592

Wagner, V., Večeřa, M., Jiménez-Alfaro, B., Pergl, J., Lenoir, J., Svenning, J., Pyšek, P., Agrillo, E., Biurrun, I., Campos, J. A., Ewald, J., Fernández-González, F., Jandt, U., Rašomavičius, V., Šilc, U., Škvorc, Ž., Vassilev, K., Wohlgemuth, T., & Chytrý, M. (2021). Alien plant invasion hotspots and invasion debt in European woodlands. *Journal of Vegetation Science, 32*(2), e13014. https://doi.org/10.1111/jvs.13014

Wagner, W. L., Herbst, D. R., & Sohmer, S. H. (1999). *Manual of the Flowering Plants of Hawai'i, Vols. 1 and 2.* University of Hawai'i and Bishop Museum Press.

Waits, A., Emelyanova, A., Oksanen, A., Abass, K., & Rautio, A. (2018). Human infectious diseases and the changing climate in the Arctic. *Environment International, 121*, 703–713. https://doi.org/10.1016/j.envint.2018.09.042

Walker, B. H., & Noy-Meir, I. (1982). Aspects of the Stability and Resilience of Savanna Ecosystems. In B. J. Huntley & B. H. Walker (Eds.), *Ecology of Tropical Savannas* (Vol. 42, pp. 556–590). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-68786-0_26

Wallace, J. M., Pavek, P. L. S., & Prather, T. S. (2015). Ecological characteristics of *Ventenata dubia* in the Intermountain Pacific Northwest. *Invasive Plant Science and Management, 8*(1), 57–71. Cambridge Core. https://doi.org/10.1614/IPSM-D-14-00034.1

Walsh, N. G., & Stajsic, V. (Eds.). (2007). *A census of the vascular plants of Victoria* (8th ed.). National Herbarium of Victoria, Royal Botanic Gardens Melbourne.

Walther, G.-R., Roques, A., Hulme, P. E., Sykes, M. T., Pyšek, P., Kühn, I., Zobel, M., Bacher, S., Botta-Dukát, Z., Bugmann, H., Czúcz, B., Dauber, J., Hickler, T., Jarošík, V., Kenis, M., Klotz, S., Minchin, D., Moora, M., Nentwig, W., … Settele, J. (2009). Alien species in a warmer world: Risks and opportunities. *Trends in Ecology & Evolution, 24*(12), 686–693. https://doi.org/10.1016/j.tree.2009.06.008

Wan, F.-H., & Yang, N.-W. (2016). Invasion and management of agricultural alien insects in China. *Annual Review of Entomology, 61*(1), 77–98. https://doi.org/10.1146/annurev-ento-010715-023916

Wan, J.-Z., Wang, C.-J., & Yu, F.-H. (2016). Risk hotspots for terrestrial plant invaders under climate change at the global scale. *Environmental Earth Sciences, 75*(12), 1012. https://doi.org/10.1007/s12665-016-5826-8

Wan, J.-Z., Zhang, Z.-X., & Wang, C.-J. (2019). Effects of ecoregional vulnerability on habitat suitability of invasive alien plants: An assessment using 13 species on a global scale. *Environmental Earth Sciences, 78*(5), 180. https://doi.org/10.1007/s12665-019-8186-3

Wang, C., Hawthorne, D., Qin, Y., Pan, X., Li, Z., & Zhu, S. (2017). Impact of climate and host availability on future distribution of Colorado potato beetle. *Scientific Reports, 7*(1), 4489. https://doi.org/10.1038/s41598-017-04607-7

Wang, C.-J., Wan, J.-Z., & Zhang, Z.-X. (2017). Expansion potential of invasive tree plants in ecoregions under climate change scenarios: An assessment of 54 species at a global scale. *Scandinavian Journal of Forest Research, 32*(8), 663–670. https://doi.org/10.1080/02827581.2017.1283049

Wang, Q., Cheng, L., Liu, J., Li, Z., Xie, S., & De Silva, S. S. (2015). Freshwater aquaculture in PR China: Trends and prospects. *Reviews in Aquaculture, 7*(4), 283–302. https://doi.org/10.1111/raq.12086

Wang, Y., & Xu, Z. (2016). Where are the Alien Species? Predictions of Global Plant Species Invasions under Current Environmental Conditions and the Human Footprint. *Polish Journal of Environmental Studies, 25*(4), 1729–1738. https://doi.org/10.15244/pjoes/62094

Wangensteen, O. S., Cebrian, E., Palacín, C., & Turon, X. (2018). Under the canopy: Community-wide effects of invasive algae in Marine Protected Areas revealed by metabarcoding. *Marine Pollution Bulletin, 127*, 54–66. https://doi.org/10.1016/j.marpolbul.2017.11.033

Ward, D. F., Beggs, J. R., Clout, M. N., Harris, R. J., & O'Connor, S. (2006). The diversity and origin of exotic ants arriving in New Zealand via human-mediated dispersal. *Diversity and Distributions, 12*(5), 601–609. https://doi.org/10.1111/j.1366-9516.2006.00270.x

Ware, C., Berge, J., Jelmert, A., Olsen, S. M., Pellissier, L., Wisz, M., Kriticos, D., Semenov, G., Kwaśniewski, S., & Alsos, I. G. (2016). Biological introduction risks from shipping in a warming Arctic. *Journal of Applied Ecology, 53*(2), 340–349. https://doi.org/10.1111/1365-2664.12566

Ware, C., Bergstrom, D. M., Müller, E., & Alsos, I. G. (2012). Humans introduce viable seeds to the Arctic on footwear. *Biological Invasions, 14*(3), 567–577. https://doi.org/10.1007/s10530-011-0098-4

Wasowicz, P., Przedpelska-Wasowicz, E. M., & Kristinsson, H. (2013). Alien vascular plants in Iceland: Diversity, spatial patterns, temporal trends, and the impact of climate change. *Flora – Morphology, Distribution, Functional Ecology of Plants, 208*(10–12), 648–673. https://doi.org/10.1016/j.flora.2013.09.009

Wasowicz, P., Sennikov, A. N., Westergaard, K. B., Spellman, K., Carlson, M., Gillespie, L. J., Saarela, J. M., Seefeldt, S. S., Bennett, B., Bay, C., Ickert-Bond, S., & Väre, H. (2020). Non-native vascular flora of the Arctic: Taxonomic richness, distribution and pathways. *Ambio, 49*(3), 693–703. https://doi.org/10.1007/s13280-019-01296-6

Watts, C. H., Clarkson, B. R., Cave, V. M., Thornburrow, D., & Thorpe, S. (2020). Invertebrate communities in a modified isolated raised bog compared to an intact raised bog in New Zealand. *Mires and Peat, 26*, 20. https://doi.org/10.19189/MaP.2019.MEH.StA.1879

Wavrek, M., Heberling, J. M., Fei, S., & Kalisz, S. (2017). Herbaceous invaders in temperate forests: A systematic review of their ecology and proposed mechanisms of invasion. *Biological Invasions, 19*(11), 3079–3097. https://doi.org/10.1007/s10530-017-1456-7

Weber, E., & Li, B. (2008). Plant invasions in China: What is to be expected in the wake of economic development? *BioScience, 58*(5), 437–444. https://doi.org/10.1641/B580511

Weber, E., Sun, S.-G., & Li, B. (2008). Invasive alien plants in China: Diversity and ecological insights. *Biological Invasions, 10*(8), 1411–1429. https://doi.org/10.1007/s10530-008-9216-3

Weldon, C., du Preez, L. H., Hyatt, A. D., Muller, R., & Speare, R. (2004). Origin of the amphibian chytrid fungus. *Emerging Infectious Diseases, 10*(12), 2100–2105. https://doi.org/10.3201/eid1012.030804

Wesche, K., Ambarlı, D., Kamp, J., Török, P., Treiber, J., & Dengler, J. (2016). The Palaearctic steppe biome: A new synthesis. *Biodiversity and Conservation, 25*(12), 2197–2231. https://doi.org/10.1007/s10531-016-1214-7

Western, L., & Juvik, J. O. (1983). Roadside plant communities on Mauna Loa, Hawaii. *Journal of Biogeography, 10*(4), 307–316. https://doi.org/10.2307/2844740

Wetterer, J. K., Wild, A. L., Suarez, A. V., Roura-Pascual, N., & Espadaler, X. (2009). Worldwide spread of the Argentine ant, *Linepithema humile* (Hymenoptera: Formicidae). *Myrmecological News, 12*, 187–194. https://sib.illinois.edu/suarez/local/suarez/uploads/2020/01/Wetterer_etal2009MyrNews.pdf

Weyl, O. L. F., Ellender, B. R., Wassermann, R. J., Truter, M., Dalu, T., Zengeya, T. A., & Smit, N. J. (2020). Alien freshwater fauna in South Africa. In *Biological invasions in South Africa* (Vol. 14, pp. 151–182). Springer. https://doi.org/10.1007/978-3-030-32394-3_6

White, R. P., Murray, S., & Rohweder, M. (2000). *Pilot analysis of global ecosystems: Grassland ecosystems.* World Resources Institute. http://pdf.wri.org/page_grasslands.pdf

Whittaker, R. J., & Fernández-Palacios, J. M. (2006). *Island Biogeography; ecology, evolution, and conservation* (121). Oxford University Press. https://doi.org/10.4000/mediterranee.6942

Wieczorek, K., & Chłond, D. (2019). The first detection of the alien species: Green-peach aphid *Myzus (Nectarosiphon) persicae* (Insecta, Hemiptera, Aphididae) in the Svalbard archipelago. *Polar Biology, 42*(10), 1947–1951. https://doi.org/10.1007/s00300-019-02562-9

Wijesundara, S. (2010). Invasive Alien Plants in Sri Lanka. In B. Marambe, P. Silva, S. Wijesundara, & N. Atapattu (Eds.), *Invasive Alien Species in Sri Lanka – Strengthening Capacity to Control Their Introduction and Spread* (pp. 27–38). Biodiversity Secretariat of the Ministry of Environment. http://envmin.nsf.gov.lk/handle/1/382

Wilkinson, M. D., Dumontier, M., Aalbersberg, Ij. J., Appleton, G., Axton, M., Baak, A., Blomberg, N., Boiten, J.-W., da Silva Santos, L. B., Bourne, P. E., Bouwman, J., Brookes, A. J., Clark, T., Crosas, M., Dillo, I., Dumon, O., Edmunds, S., Evelo, C. T., Finkers, R., … Mons, B. (2016). The FAIR Guiding Principles for scientific data management and stewardship. *Scientific Data, 3*(1), 160018. https://doi.org/10.1038/sdata.2016.18

Willby, N. J. (2007). Managing invasive aquatic plants: Problems and prospects. *Aquatic Conservation: Marine and Freshwater Ecosystems, 17*(7), 659–665. https://doi.org/10.1002/aqc.913

Willette, D. A., Chalifour, J., Debrot, A. O. D., Engel, M. S., Miller, J., Oxenford, H. A., Short, F. T., Steiner, S. C. C., & Védie, F. (2014). Continued expansion of the trans-Atlantic invasive marine angiosperm *Halophila stipulacea* in the Eastern Caribbean. *Aquatic Botany, 112*, 98–102. https://doi.org/10.1016/j.aquabot.2013.10.001

Williams, D. G., & Baruch, Z. (2000). African grass invasion in the Americas: Ecosystem consequences and the role of ecophysiology. *Biological Invasions, 2*(2), 123–140. https://doi.org/10.1023/A:1010040524588

Williams, M. C., & Wardle, G. M. (2005). The invasion of two native Eucalypt forests by *Pinus radiata* in the Blue Mountains, New South Wales, Australia. *Biological Conservation, 125*(1), 55–64. https://doi.org/10.1016/j.biocon.2005.03.011

Williams, S. L., & Smith, J. E. (2007). A global review of the distribution, taxonomy, and impacts of introduced seaweeds. *Annual Review of Ecology, Evolution, and Systematics, 38*(1), 327–359. https://doi.org/10.1146/annurev.ecolsys.38.091206.095543

Williamson, M., & Fitter, A. (1996). The varying success of invaders. *Ecology, 77*(6), 1661–1666. https://doi.org/10.2307/2265769

Wilson, J. R. U., Richardson, D. M., Rouget, M., Procheş, Ş., Amis, M. A., Henderson, L., & Thuiller, W. (2007). Residence time and potential range: Crucial considerations in modelling plant invasions. *Diversity and Distributions, 13*(1), 11–22. https://doi.org/10.1111/j.1366-9516.2006.00302.x

Winters, G., Beer, S., Willette, D. A., Viana, I. G., Chiquillo, K. L., Beca-Carretero, P., Villamayor, B., Azcárate-García, T., Shem-Tov, R., Mwabvu, B., Migliore, L., Rotini, A., Oscar, M. A., Belmaker, J., Gamliel, I., Alexandre, A., Engelen, A. H., Procaccini, G., & Rilov, G. (2020). The tropical seagrass *Halophila stipulacea*: Reviewing what we know from its native and invasive habitats, alongside identifying knowledge gaps. *Frontiers in Marine Science, 7*, 300. https://doi.org/10.3389/fmars.2020.00300

Witt, A. B. R., Kiambi, S., Beale, T., & Van Wilgen, B. W. (2017). A preliminary assessment of the extent and potential impacts of alien plant invasions in the Serengeti-Mara ecosystem, East Africa. *Koedoe, 59*(1), a1426. https://doi.org/10.4102/koedoe.v59i1.1426

Witt, A. B. R., Nunda, W., Beale, T., & Kriticos, D. J. (2020). A preliminary assessment of the presence and distribution of invasive and potentially invasive alien plant species in Laikipia County, Kenya, a biodiversity hotspot. *Koedoe, 62*(1), 1–10. https://doi.org/10.4102/koedoe.v62i1.1605

Witt, A. B. R., Shackleton, R. T., Beale, T., Nunda, W., & Van Wilgen, B. W. (2019). Distribution of invasive alien *Tithonia* (Asteraceae) species in eastern and southern Africa and the socio-ecological impacts of *T. diversifolia* in Zambia. *Bothalia, 49*(1), a2356. https://doi.org/10.4102/abc.v49i1.2356

Wittmer, M. H. O. M., Auerswald, K., Bai, Y., Schäufele, R., & Schnyder, H. (2010). Changes in the abundance of C3/C4 species of Inner Mongolia grassland: Evidence from isotopic composition of soil and vegetation. *Global Change Biology, 16*(2), 605–616. https://doi.org/10.1111/j.1365-2486.2009.02033.x

Woinarski, J. C. Z., Burbidge, A. A., & Harrison, P. L. (2015). Ongoing unraveling of a continental fauna: Decline and extinction of Australian mammals since European settlement. *Proceedings of the National Academy of Sciences, 112*(15), 4531–4540. https://doi.org/10.1073/pnas.1417301112

Wolff, W. J. (2005). Non-indigenous marine and estuarine species in The Netherlands. *Zoologische Mededelingen, Leiden, 79*(1), 1–116.

Wood, A. R. (2017). Fungi and invasions in South Africa. *Bothalia, 47*(2), a2124. https://doi.org/10.4102/abc.v47i2.2124

Woodford, D. J., & Impson, N. D. (2004). A preliminary assessment of the impact of alien rainbow trout *(Oncorhynchus mykiss)* on indigenous fishes of the upper Berg River, Western Cape Province, South Africa. *African Journal of Aquatic Science, 29*(1), 107–111. https://doi.org/10.2989/16085910409503799

Woodham-Smith, C. (1962). *The great hunger: Ireland 1845 – 1849.* Penguin Books.

Woods, T. M., Strakosh, S. C., Nepal, M. P., Chakrabarti, S., Simpson, N. B., Mayfield, M. H., & Ferguson, C. J. (2005). Introduced species in Kansas: Floristic changes and patterns of collection based on an historical herbarium. *SIDA, Contributions to Botany, 21*(3), 1695–1725. JSTOR. http://www.jstor.org/stable/41968449

World Spider Catalog. (2017). *World Spider Catalog.* Natural History Museum Bern. https://doi.org/10.24436/2

Wroe, S., Field, J., & Grayson, D. K. (2006). Megafaunal extinction: Climate, humans and assumptions. *Trends in Ecology & Evolution, 21*(2), 61–62. https://doi.org/10.1016/j.tree.2005.11.012

Wu, S.-H., Yang, T. Y. A., Teng, Y.-C., Chang, C.-Y., Yang, K.-C., & Hsieh, C.-F. (2010). Insights of the latest naturalized flora of Taiwan: Change in the past eight years. *Taiwania, 55*(2), 139–159. https://doi.org/10.6165/tai.2010.55(2).139

WWF. (2018). *Living Planet Report—2018: Aiming Higher* (M. Grooten & R. E. A. Almond, Eds.). World Wide Fund for Nature.

Wyse, S. V., Wilmshurst, J. M., Burns, B. R., & Perry, G. L. W. (2018). New Zealand forest dynamics: A review of past and present vegetation responses to disturbance, and development of conceptual forest models. *New Zealand Journal of Ecology, 42*(2), 87–106. https://doi.org/10.20417/nzjecol.42.18

Xiong, W., Shen, C., Wu, Z., Lu, H., & Yan, Y. (2017). A brief overview of known introductions of non-native marine and coastal species into China. *Aquatic Invasions, 12*(1), 109–115. https://doi.org/10.3391/ai.2017.12.1.11

Xiong, W., Sui, X., Liang, S.-H., & Chen, Y. (2015). Non-native freshwater fish species in China. *Reviews in Fish Biology and Fisheries, 25*(4), 651–687. https://doi.org/10.1007/s11160-015-9396-8

Xu, H. G., & Qiang, S. (2018). *China's invasive alien species (revised edition)*. Science Press.

Xu, H., Qiang, S., Genovesi, P., Ding, H., Wu, J., Meng, L., Han, Z., Miao, J., Hu, B., Guo, J., Sun, H., Huang, C., Lei, J., Le, Z., Zhang, X., He, S., Wu, Y., Zheng, Z., Chen, L., … Pyšek, P. (2012). An inventory of invasive alien species in China. *NeoBiota, 15*, 1–26. https://doi.org/10.3897/neobiota.15.3575

Xun, Z., Bai, L., Qu, B., Xu, Y., Li, G., Zhan, Z., & Shi, J. (2017). Effect of nitrogen treatments on growth of the invasive plant *Xanthium strumarium*, the native plant *Xanthium sibiricum*, and their reciprocal crosses. *Acta Prataculturae Sinica, 26*(5), 51–61. https://doi.org/10.11686/cyxb2016217

Yamada, S. B., & Mansour, R. A. (1987). Growth inhibition of native *Littorina saxatilis* (Olivi) by introduced *L. littorea* (L.). *Journal of Experimental Marine Biology and Ecology, 105*(2–3), 187–196. https://doi.org/10.1016/0022-0981(87)90171-7

Yamaguchi, R., Yamanaka, T., & Liebhold, A. M. (2019). Consequences of hybridization during invasion on establishment success. *Theoretical Ecology, 12*(2), 197–205. https://doi.org/10.1007/s12080-019-0415-6

Yamanaka, T., Morimoto, N., Nishida, G. M., Kiritani, K., Moriya, S., & Liebhold, A. M. (2015). Comparison of insect invasions in North America, Japan and their Islands. *Biological Invasions, 17*(10), 3049–3061. https://doi.org/10.1007/s10530-015-0935-y

Yang, M., Lu, Z., Liu, X., De Wulf, R., Hens, L., & Ou, X. (2018). Association of non-native plant species with recreational roads in a National Park in the Eastern Himalayas, China. *Mountain Research and Development, 38*(1), 53–62. https://doi.org/10.1659/MRD-JOURNAL-D-17-00012.1

Yang, M. Y., Macaya, E. C., & Kim, M. S. (2015). Molecular evidence for verifying the distribution of *Chondracanthus chamissoi* and *C. teedei* (Gigartinaceae, Rhodophyta). *Botanica Marina, 58*(2), 103–113. https://doi.org/10.1515/bot-2015-0011

Yang, Q.-Q., Liu, S.-W., He, C., & Yu, X.-P. (2018). Distribution and the origin of invasive apple snails, *Pomacea canaliculata* and *P. maculata* (Gastropoda: Ampullariidae) in China. *Scientific Reports, 8*(1), 1185. https://doi.org/10.1038/s41598-017-19000-7

Yap, T. A., Nguyen, N. T., Serr, M., Shepack, A., & Vredenburg, V. T. (2017). *Batrachochytrium salamandrivorans* and the risk of a second amphibian pandemic. *EcoHealth, 14*(4), 851–864. https://doi.org/10.1007/s10393-017-1278-1

Yazlık, A., Pergl, J., & Pyšek, P. (2018). Impact of alien plants in Turkey assessed by the Generic Impact Scoring System. *NeoBiota, 39*, 31–51. https://doi.org/10.3897/neobiota.39.23598

Yonow, T., Hattingh, V., & de Villiers, M. (2013). CLIMEX modelling of the potential global distribution of the citrus black spot disease caused by *Guignardia citricarpa* and the risk posed to Europe. *Crop Protection, 44*, 18–28. https://doi.org/10.1016/j.cropro.2012.10.006

Yoshida, K., Schuenemann, V. J., Cano, L. M., Pais, M., Mishra, B., Sharma, R., Lanz, C., Martin, F. N., Kamoun, S., Krause, J., Thines, M., Weigel, D., & Burbano, H. A. (2013). The rise and fall of the *Phytophthora infestans* lineage that triggered the Irish potato famine. *eLife, 2*, e00731. https://doi.org/10.7554/eLife.00731

Yuldashov, M. A. (2018). Introduction of alien fish species to waterbodies of Uzbekistan. *International Journal of Science and Research, 7*(1), 1213–1219. https://doi.org/10.21275/ART20179585

Yuma, M., Hosoya, K., & Nagata, Y. (1998). Distribution of the freshwater fishes of Japan: An historical overview. *Environmental Biology of Fishes, 52*(1–3), 97–124. https://doi.org/10.1023/A:1007451417556

Zaitsev, Y. P. (1992). Recent changes in the trophic structure of the Black Sea. *Fisheries Oceanography, 1*(2), 180–189. https://doi.org/10.1111/j.1365-2419.1992.tb00036.x

Zalba, S. M., & Amodeo, M. R. (2015). Site-specific reproductive performance of an invasive fleshy-fruited tree in remnant grasslands of the Argentinean Pampas. *Weed Research, 55*(6), 546–554. https://doi.org/10.1111/wre.12171

Zalba, S. M., & Loydi, A. (2014). The influence of feral horses dung piles on surrounding vegetation. *Management of Biological Invasions, 5*(1), 73–79. https://doi.org/10.3391/mbi.2014.5.1.07

Zedler, J. B., & Kercher, S. (2004). Causes and Consequences of Invasive Plants in Wetlands: Opportunities, Opportunists, and Outcomes. *Critical Reviews in Plant Sciences, 23*(5), 431–452. https://doi.org/10.1080/07352680490514673

Zefferman, E., Stevens, J. T., Charles, G. K., Dunbar-Irwin, M., Emam, T., Fick, S., Morales, L. V., Wolf, K. M., Young, D. J. N., & Young, T. P. (2015). Plant communities in harsh sites are less invaded: A summary of observations and proposed explanations. *AoB PLANTS, 7*, plv056. https://doi.org/10.1093/aobpla/plv056

Zemanova, M. A., Broennimann, O., Guisan, A., Knop, E., & Heckel, G. (2018). Slimy invasion: Climatic niche and current and future biogeography of *Arion* slug invaders. *Diversity and Distributions, 24*(11), 1627–1640. https://doi.org/10.1111/ddi.12789

Zenetos, A., & Galanidi, M. (2020). Mediterranean non indigenous species at the start of the 2020s: Recent changes. *Marine Biodiversity Records, 13*(1), 10. https://doi.org/10.1186/s41200-020-00191-4

Zengeya, T. A., Kumschick, S., Weyl, O. L. F., & van Wilgen, B. W. (2020). An evaluation of the impacts of alien species on biodiversity in South Africa using different assessment methods. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 489–512). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_17

Zengeya, T. A., & Wilson, J. R. (2020). *The status of biological invasions and their management in South Africa in 2019* (Final Report) [Final Report]. Zenodo. https://doi.org/10.5281/ZENODO.3947613

Zenni, R. D. (2015). The naturalized flora of Brazil: A step towards identifying future invasive non-native species. *Rodriguésia, 66*(4), 1137–1144. https://doi.org/10.1590/2175-7860201566413

Zenni, R. D., Herrera, I., de Sá Dechoum, M., Ziller, S. R., de Matos, A. C. L., Núñez, C. I., Núñez, M. A., & Pauchard, A. (2022). Plant Invasions in South America. In D. R. Clements, M. K. Upadhyaya, S. Joshi, & A. Shrestha (Eds.), *Global Plant Invasions* (pp. 187–208). Springer International Publishing. https://doi.org/10.1007/978-3-030-89684-3_9

Zenni, R. D., & Nuñez, M. A. (2013). The elephant in the room: The role of failed invasions in understanding invasion biology. *Oikos, 122*(6), 801–815. https://doi.org/10.1111/j.1600-0706.2012.00254.x

Zenni, R. D., & Ziller, S. R. (2011). An overview of invasive plants in Brazil. *Revista Brasileira de Botânica, 34*(3), 431–446. https://doi.org/10.1590/S0100-84042011000300016

Zhu, G.-L., Tang, Y.-Y., Limpanont, Y., Wu, Z.-D., Li, J., & Lv, Z.-Y. (2019). Zoonotic parasites carried by invasive alien species in China. *Infectious Diseases of Poverty, 8*(1), 2. https://doi.org/10.1186/s40249-018-0512-6

Zhu, Z.-X., Roeder, M., Xie, J., Nizamani, M. M., Friedman, C. R., & Wang, H.-F. (2019). Plant taxonomic richness and phylogenetic diversity across different cities in China. *Urban Forestry & Urban Greening, 39*, 55–66. https://doi.org/10.1016/j.ufug.2019.02.004

Zieritz, A., Gallardo, B., Baker, S. J., Britton, J. R., van Valkenburg, J. L. C.

H., Verreycken, H., & Aldridge, D. C. (2017). Changes in pathways and vectors of biological invasions in Northwest Europe. *Biological Invasions, 19*(1), 269–282. https://doi.org/10.1007/s10530-016-1278-z

Zufiaurre, E., Abba, A., Bilenca, D., & Codesido, M. (2016). Role of landscape elements on recent distributional expansion of European Starlings *(Sturnus vulgaris)* in agroecosystems of the Pampas, Argentina. *The Wilson Journal of Ornithology, 128*(2), 306–313. https://doi.org/10.1676/wils-128-02-306-313.1

Zug, G. R. (2013). *Reptiles and amphibians of the Pacific Islands: A comprehensive guide* (1st ed.). University of California Press. https://www.jstor.org/stable/10.1525/j.ctt2jcc1n

Zvyagintsev, A. Yu., Radashevsky, V. I., Ivin, V. V., Kashin, I. A., & Gorodkov, A. N. (2011). Nonindigenous species in the Far Eastern seas of Russia. *Russian Journal of Biological Invasions, 2*(2–3), 164–182. https://doi.org/10.1134/S2075111711030210

Chapter **3**

DRIVERS AFFECTING BIOLOGICAL INVASIONS[1]

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DRIVERS AFFECTING BIOLOGICAL INVASIONS

EXECUTIVE SUMMARY

1 The IPBES conceptual framework, which classes drivers as either indirect or direct with respect to their impact on nature, can be adapted to address biological invasions *(well established)* {3.1.3}. The IPBES conceptual framework distinguishes between direct drivers of change in natureand indirect drivers, which are societal factors that act on biodiversity and ecosystems through influencing one or more direct drivers {3.1.3}. Biological invasions are facilitated by a broad range of direct drivers {3.3}. As invasive alien species are both intentionally and unintentionally moved by people, some indirect drivers of change in nature, such as trade or travel, can facilitate the transport and introduction of invasive alien species to new regions *(well established)* {3.2.3.1, 3.2.3.2, 3.2.3.3, 3.2.3.4}. Since invasive alien species are part of and interact with nature, changes in the biosphere can also facilitate biological invasions, for example, biodiversity loss can lead to reduced biotic resistance of ecosystems to invasive alien species, and invasive alien species can facilitate the establishment and spread of other alien species *(well established)* {3.3.5, 3.4.2}. Natural drivers, such as tsunamis or hurricanes, are known to facilitate the establishment and further spread of invasive alien species *(well established)* {3.4.1}. By incorporating aspects specific to invasive alien species within the IPBES conceptual framework, this chapter allows a comprehensive assessment of all factors influencing biological invasions **(Figure 3.3)**.

2 Indirect and direct drivers of change in nature play significant but varying roles across all stages of the invasion process *(well established)* {3.1.4, 3.2, 3.3, 3.6.2}. Indirect drivers such as sociocultural norms (particularly human values), demography (human population and migration), economic aspects (especially trade and travel), science and technology (including research) and governance (including unintended consequences of policies that inadvertently facilitate biological invasions) tend to play a stronger role in the transport and introduction stages of the invasion process *(well established)* {3.2, 3.6.2}. In contrast, the five broad classes of direct drivers examined, land- and sea-use changes (resulting from agriculture, forestry and aquaculture), direct exploitation of natural resources (mining and species harvest), pollution (eutrophication and marine plastics), climate change and invasive alien species are all more influential in the establishment and spread of invasive alien species *(established but incomplete)* {3.3, 3.6.2}.

3 The magnitude of most drivers of change in nature have increased significantly since 1950, contributing to the increase of invasive alien species globally *(well established)* {3.1.1}. Recent decades have been characterized by increases in global trade and travel, human population size and urbanization, land- and sea-use change, habitat and biodiversity loss and degradation, direct exploitation of natural resources and pollution, and global temperatures along with shifts in precipitation patterns *(well established)* {3.1.1} **(Figure 3.1)**. This sustained growth in many of the key drivers affecting the transport, introduction, establishment and spread of invasive alien species underlies recent increases in the rates of introduction, establishment and spread of invasive alien species globally *(well established)* {3.2.1, 3.6.2}.

4 International trade is the most significant driver responsible for the transport and introduction of invasive alien species across the globe *(well established)* {3.2.3.1, 3.6.2}. International trade, primarily maritime transport of commodities, has been responsible for the transport and introduction of numerous invasive alien species in both terrestrial and aquatic biomes *(well established)* {3.2.3.1}. Invasive alien species can be the commodity traded, such as ornamental plants, contaminants of commodities, such as weed seeds in grain shipments, or stowaways on shipping containers or vessels, such as biofoulants *(well established)* {3.2.3}. Historically, intentional as well as unintentional introductions through the release or escape of plants, animals and microbial organisms from agriculture, aquaculture, forestry, fisheries and non-commercial uses, have resulted in the establishment and spread of alien species in terrestrial, aquatic and marine ecosystems worldwide *(well established)* {3.3.1.1}. Biofouling and ballast water discharges have had a major influence on biological invasions in coastal marine ecosystems *(well established)* {3.2.3.1}. International trade also influences other drivers of change in nature that facilitate biological invasions, for example by promoting urbanization around major trade ports, driving land- and sea-use changes and direct exploitation of natural resources to meet international market demands, and increasing atmospheric and aquatic pollution *(well established)* {3.2.3.1, 3.2.2.3, 3.3.1, 3.3.2, 3.3.3}.

5 Land-use changes are the most significant drivers accelerating the establishment and spread of invasive alien species *(well established)* {3.3.1, 3.6.2}. Land-use changes are major drivers facilitating invasive alien species by providing opportunities for colonization, establishment and spread of alien species in both terrestrial and coastal environments worldwide *(well established)* {3.3.1}. Land-use changes related to food, fodder and biomass production facilitate the biological invasion process through the replacement of native ecosystems by monocultures of introduced crops and livestock and through intensification and changes in disturbance regimes *(well established)* {3.3.1.1, 3.3.1.2, 3.3.1.5, 3.3.1.6}. Land-use changes related to industry, infrastructure and urban development facilitate biological invasions through creation of corridors and artificial surfaces in terrestrial and coastal environments and more generally through landscape degradation *(well established)* {3.3.1.3, 3.3.1.4, 3.3.1.6}. Fragmented and degraded ecosystems are often vulnerable to colonization and spread by generalist invasive alien species *(well established)* {3.3.1.2, 3.3.1.3}.

6 **Many of the drivers known to negatively impact nature and nature's contributions to people also facilitate the introduction and spread of invasive alien species, potentially causing positive feedback loops *(established but incomplete)* {3.5}.** Increasing and expanding trade and rapid population and economic growth are global phenomena that facilitate the transport and introduction of invasive alien species worldwide, while increasing urbanization, land- and sea-use changes, pollution, ecosystem degradation and biodiversity loss are changes which again facilitate the establishment and spread of invasive alien species *(established but incomplete)* {3.2.2, 3.2.3, 3.3.1, 3.3.3, 3.4.1, 3.4.2, 3.5.3}. Such positive feedback loops between drivers remain poorly understood but are critical to understanding and addressing complex spatial patterns and temporal dynamics in biological invasions *(established but incomplete)* {3.1.5, 3.2.1, 3.2.3.1, 3.2.3.2, 3.3.4, 3.5, 3.6.1}.

7 **Historically important drivers of change in nature such as trade, land-use change and direct exploitation of natural resources remain major causes of invasive alien species introduction and spread but the role of climate change and biodiversity loss will increasingly shape future global trends in invasive alien species, potentially with a significant temporal lag *(established but incomplete)* {3.2.3, 3.3.1, 3.3.2, 3.3.4, 3.4.2}.** While some countries are moving away from intentional introductions of alien species for uses in agriculture, aquaculture, forestry, horticulture, fishing, hunting and ornamental purposes, other countries do not effectively regulate and manage the use of invasive alien species for these purposes, resulting in sustained or increased rates of introduction and spread in affected regions *(established but incomplete)* {3.2.5, 3.2.3.2, 3.3.1.1}. The increasing role of climate change and biodiversity loss in facilitating the establishment and spread of invasive alien species is indicative that past patterns of biological invasions may not be a good guide to future patterns *(well established)* {3.3.4, 3.4.2}. Furthermore, there will be a vast legacy of future invasions (invasion debt) caused by significant time lags in the response of invasive alien species to drivers of change in nature, and the ongoing intensification of many drivers are responsible for increases in this legacy *(established but incomplete)* {3.1.1, 3.1.5}.

8 **Despite major inequalities in wealth worldwide, economic growth facilitates biological invasions in both developed and developing countries *(established but incomplete)* {3.2.3}.** Countries with high levels of consumption tend to expedite the introduction and establishment of alien species, and the cumulative build-up of assets, which support greater consumption, may lead to more immediate increases in numbers of alien species *(established but incomplete)* {3.2.3.6}. Poverty and marginalization created by economic inequality within and among countries may indirectly drive the introduction, establishment and spread of invasive alien species *(well established)* {3.2.3.6}. For those countries with a lower level of wealth, trends suggest that as economies grow and larger asset bases are built, so the risk of invasive alien species introductions might increase *(established but incomplete)* {3.2.3.6}. Risks may also be exacerbated where the route to economic growth and poverty reduction encourages the development of economic sectors based around invasive alien species *(established but incomplete)* {3.2.3.6}. This appears to be a major issue for Indigenous Peoples and local communities who in some cases may have few options but to use invasive alien species for food, fibre and also medicines *(established but incomplete)* {3.2.3.6}.

9 **Many Indigenous Peoples and local communities have a good and holistic understanding of the drivers facilitating invasive alien species on their lands *(well established)* {3.2, 3.3, 3.5} (Box 3.6, Box 3.15).** Indigenous Peoples and local communities point to how policies, governance and institutions aimed at improving livelihoods and the environment may inadvertently cause the introduction of invasive alien species *(established but incomplete)* {3.2.5, 3.2.3.6, 3.3.1.13} **(Box 3.6, Box 3.15)**. For example, they report that promotion of alien species for food, fibre, income generation, or medicinal purposes may act as a driver facilitating biological invasions *(well established)*, and such invasions can be especially facilitated in situations where native biodiversity, including species they traditionally depended on for these benefits, have declined *(established but incomplete)* {3.2.3.6, 3.2.5}. In some cases, Indigenous Peoples and local communities observe that urban areas or anthropogenic corridors act as sources of further spread of invasive species into their lands *(established but incomplete)* {3.2.2, 3.3.1.7}, and they also recognize land-abandonment, sometimes coupled with natural drivers or climate extremes, as responsible for the spread of alien species *(established but incomplete)*

{3.3.1.5.1}. Indigenous Peoples and local communities are well aware that drivers interact in complex ways to drive biological invasions *(well established)* {3.5}. Indigenous Peoples and local communities also identify challenges of land tenure and access rights as significant factors limiting the extent to which they can address invasive alien species on their lands *(established but incomplete)* {3.2.5}. Overall, Indigenous Peoples and local communities broadly align in understanding of the relative importance of drivers and trends in invasive alien species with reports from the scientific literature *(established but incomplete)* **(Box 3.15)**.

10 Few drivers act in isolation, and interactive effects appear to be crucially important, but few studies have examined the interactive effects of several co-occurring drivers in facilitating invasive alien species *(established but incomplete)* {3.1.5, 3.5, 3.6.1, 3.6.3}. There are potentially many multiplicative interactions among drivers that are likely to lead to unprecedented invasion scenarios *(established but incomplete)* {3.1.6, 3.5, 3.6.3}. Yet fewer than 5 per cent of studies examining drivers of biological invasions addressed more than one driver *(well established)* {3.1.6, 3.6.1}. Stakeholders, including decision makers, are currently inadequately prepared to address and react to unexpected consequences arising from additive or multiplicative effects of several drivers on the transport, introduction, establishment and spread of invasive alien species *(established but incomplete)* {3.5, 3.6.3}.

11 Knowledge is biased towards only a subset of drivers, and less is understood regarding how indirect drivers of change in nature influence biological invasions compared to direct drivers *(well established)* {3.6.1}. While this chapter summarizes the available evidence of the role of each direct and indirect driver on biological invasions, the underlying knowledge base is biased *(well established)* {3.6.1}. Most of the recent research addressing the role of drivers in facilitating biological invasions has focused on a subset of drivers, especially economic drivers such as trade and transport, climate change and land-use change *(well established)* {3.6.1}. The importance of sociocultural values and unintended consequences of governance, policy and insitutions in shaping biological invasion remains understudied *(well established)* {3.6.1, 3.6.3}.

12 Biases in the availability of data on how drivers of change in nature influence biological invasions highlight that causal factors are most poorly understood for regions potentially most exposed to increasing risks from invasive alien species *(well established)* {3.6.1}. This chapter examines a variety of sources, including Indigenous and local knowledge, and explicitly includes evidence of biological invasions across multiple geographic regions, taxonomic groups and biomes {3.1.2, 3.6.1}. Nevertheless, the evidence base for how drivers influence biological invasions is largely drawn from developed countries, particularly in Europe, North America and Oceania, terrestrial temperate ecosystems and plants, and there is a lack of information for polar regions and developing countries, especially sub-Saharan Africa, tropical Asia and South America, marine systems and other taxonomic groups *(well established)* {3.6.1, 3.6.3}.

3.1 INTRODUCTION

The concept of direct and indirect drivers of change in nature has been a cornerstone in all the assessments led by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) to date (Díaz *et al.*, 2015; IPBES, 2016a, 2018f, 2018e, 2018c, 2018d, 2019; Nelson *et al.*, 2006), and the intention in this chapter is not to repeat past material pertaining to the status and trends in the drivers, but to synthesize information on the role of drivers of change in nature in affecting the biological invasion process. Chapter 3 therefore focuses on identifying how different drivers of change in nature affect the transport, introduction and establishment of invasive alien species (**Glossary**; **Box 3.1**). Chapter 3 builds on the status and trends of alien species, and the subset of these termed invasive alien species, documented in **Chapter 2**, with a more in-depth focus on establishing the drivers behind these patterns. The information provided in Chapter 3 contributes to the understanding of the underlying causes of the increase in invasive alien species globally (**Chapter 2**), the impacts of invasive alien species on nature, nature's contributions to people and good quality of life (**Glossary**; **Chapter 4**) and underpins management actions (**Glossary**; **Chapter 5**) and policy options for the prevention and control of invasive alien species and their impacts (**Glossary**; **Chapter 6**).

3.1.1 Setting the scene: increasing global trends in drivers of change in nature

The size and environmental footprint of the world's human population has grown dramatically over the past two centuries, with rates of change accelerating over the past few decades (Steffen *et al.*, 2015). This "great acceleration" (Steffen *et al.*, 2015) can be discerned across a majority of direct and indirect drivers of change in nature, which are of relevance to the increasing trends in the number and abundance of invasive alien species globally (**Chapter 2**). The number of people in the world has grown from 3.7 billion in 1970 to an estimated 7.7 billion in 2019 **(Figure 3.1)**, and while population growth is slowing, a global population of 10 billion may be reached by 2050 (UNEP, 2019; United Nations *et al.*, 2019). An increasing proportion of the global population is living in urban

Box 3 1 **Rationale of the chapter.**

Chapter 3 provides an analysis and synthesis of how direct and indirect anthropogenic drivers of change in nature, along with natural drivers and biodiversity loss itself, are responsible for the transport, introduction, establishment and spread of invasive alien species. The chapter first outlines the conceptual and analytical framework and approaches, then synthesizes the evidence for the role each driver plays across the biological invasion process (**Glossary**), before synthesizing the knowledge and identifying data gaps. Invasive alien species are one of five major classes of direct drivers of change in nature identified by the Global Assessment of Biodiversity and Ecosystem Services (IPBES, 2019), and are the theme of this assessment. Unlike other drivers, invasive alien species are both a direct driver of change in nature and they are integral parts of nature. As a consequence, invasive alien species can be directly affected by drivers that are classified as indirect drivers of change in nature, and invasive alien species along with biodiversity loss can facilitate invasion by other alien species.

Guiding questions:

- What are the main direct and indirect drivers responsible for the introduction, spread, abundance and dynamics of invasive alien species for each invasion stage and taxon?
- How rapidly are potential drivers changing compared to the last 30 years and which drivers are changing most rapidly?

Keywords:
Indirect drivers of change in nature, direct drivers of change in nature, invasive alien species, demographic drivers, economic drivers, science and technological drivers, policies, governance and institutions as drivers, land- and sea-use change, natural resource extraction, pollution, climate change, natural drivers, biodiversity loss, ecosystem resilience, interacting drivers, Indigenous Peoples and local communities.

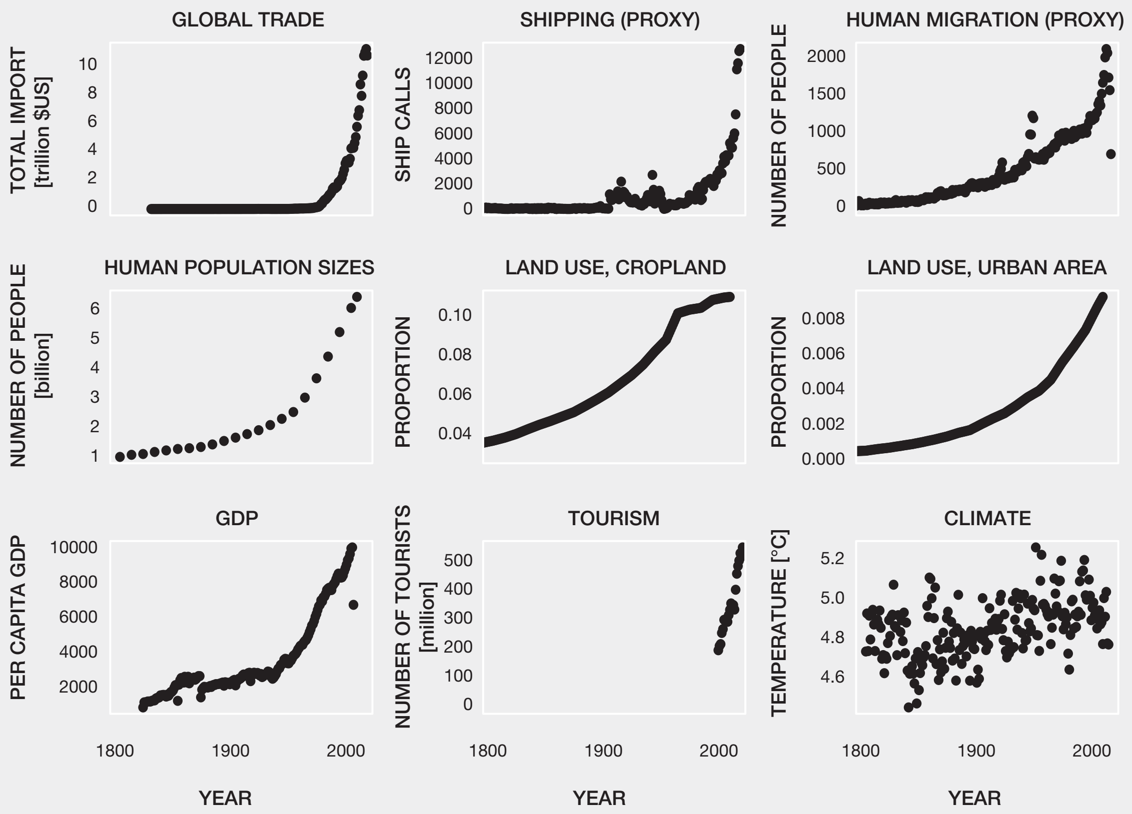


Figure 3 1 **Trends in a selection of drivers and correlates of biological invasions.**

Panels show temporal trends for some of the main drivers and correlates of biological invasions averaged globally. A data management report for this figure is available at https://doi.org/10.5281/zenodo.7615582 areas – the total area of urban settlements has grown by approximately 2.5 times since 1970, accounting for 7.6 per cent of the global land area and housing 3.5 billion people in 2015 (Shukla *et al.*, 2019; UNEP, 2019). Human migration and travel are also increasing **(Figure 3.1)**. Meanwhile, the global economy has grown nearly fivefold over the last 50 years **(Figure 3.1)**, a growth that is projected to continue. This economic growth has been fuelled by an increase in global primary energy production of more than 270 per cent over the same period, of which fossil fuels still contribute more than 80 per cent (IEA, 2020). Consumption has tripled and global trade grown nearly tenfold in the last 50 years **(Figure 3.1)**, with shifting patterns of consumption and production across regions, and increases in transport of goods and people (**Figure 3.1**; IPBES, 2019).

These global population and economic drivers are having dramatic impacts on our lands and seas (IPBES, 2022c). Close to 75 per cent of ice-free land areas and 60 per cent of the oceans are significantly impacted by people, and agricultural crop production has increased by about 300 per cent since 1970, with crops now occupying half of the habitable land on Earth (IPBES, 2019). Of the more than 50,000 wild species harvested for use as food, energy, medicine, materials, income generation, or other uses globally, only 34 per cent are used sustainably (IPBES, 2022c). Water extraction, predominantly for irrigation of agricultural crops, grew by nearly 65 per cent from 1970 to 2010, and over the same period mining of metal ores increased by three and a half times and mining for sand, gravel and clay increased by nearly five times (IRP, 2019). Approximately 60 billion tonnes of renewable and non-renewable resources are now extracted globally every year, having nearly doubled since 1980 (IPBES, 2019). Up to 400 million tonnes of heavy metals, solvents, toxic sludge and other industrial wastes are dumped annually into the

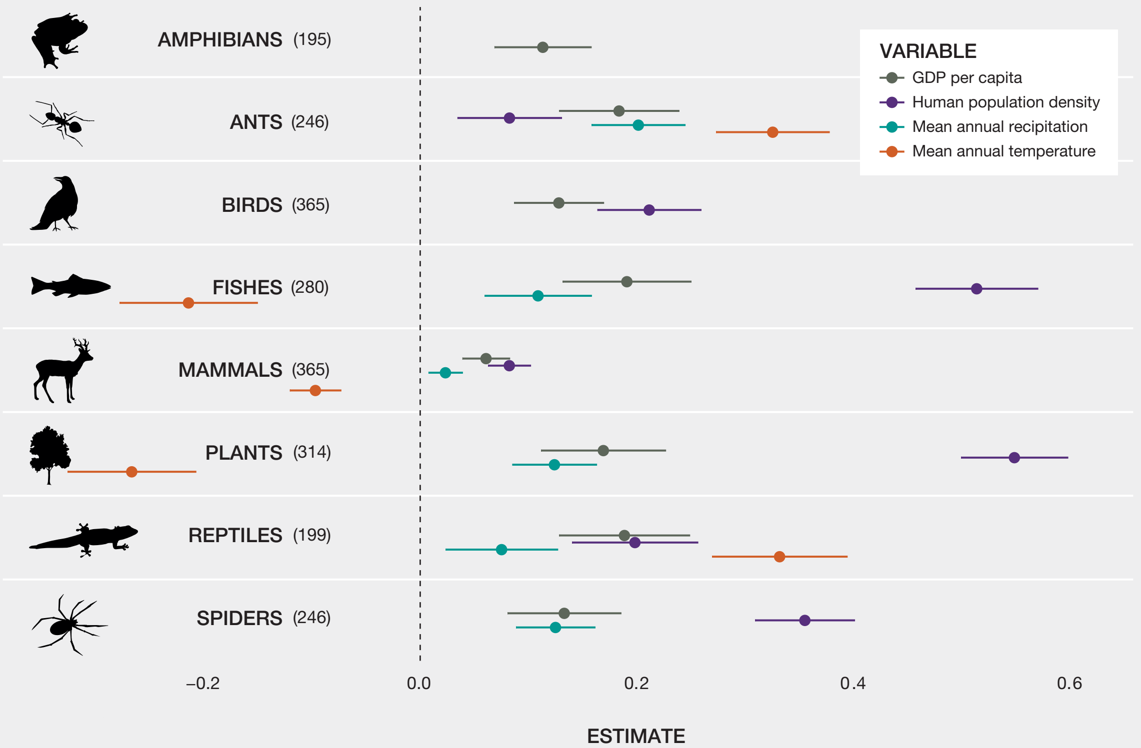


Figure 3 2 **Estimated effects of different factors on established alien species richness across eight taxonomic groups: amphibians, ants, birds, freshwater fishes, mammals, vascular plants, reptiles and spiders (from top to bottom).**

The results of linear mixed models indicate the effects of different factors (GDP per capita; human population density; mean annual precipitation; and mean annual temperature) on alien species (**Glossary**) richness within eight taxonomic groups across 423 mainland regions. Number in parentheses are numbers of regions included per taxonomic group. Overall, taxonomic groups respond differently to the effects of climate and gross domestic product (GDP) per capita, but human population density is consistently among the best predictors with especially high effects for fish, plants and spiders. Estimates (±1 standard error) – represented by dots and lines – of effects were obtained from linear mixed-effects models of ln (species richness + 1), with subcontinental regions nested within continents as random effects. Adapted from Dawson *et al.* (2017) https://doi.org/10.1038/s41559-017-0186, under license CC BY 4.0.

world's waters (UNEP, 2019; IPBES 2019). Nitrogen fluxes to aquatic ecosystems (**Glossary**) have increased up to 20-fold in the last decade (IPBES, 2019). Marine plastic pollution has increased tenfold since 1980, and is found in all oceans at all depths, concentrating in ocean currents (UNEP, 2019).

Accelerating human impacts are changing the Earth's ecosystems and climate at unprecedented rates, to the extent that they are now dominating Earth system processes (IPBES, 2019). Climate has warmed by 1.1°C on average, and is projected to reach at least 1.5°C within the next three decades, and climate change is contributing to changed precipitation patterns, sea level rise, increasing fire risk (**Glossary**) and a higher frequency of extreme events in many regions (IPCC, 2021, 2022). Ecosystems are degrading at unprecedented rates, with climate change exacerbating other threats (Pörtner *et al.*, 2021; **Chapter 4**, **Box 4.5**). This degradation of biodiversity and ecosystems is impacting ecosystem functioning and harming nature's ability to support human well-being (IPBES, 2019, 2022c; IPCC, 2022). A majority of these direct and indirect drivers of change in nature are affecting, and often facilitating, invasive alien species, which are increasing at accelerating rates globally (**Chapter 2**, **Figure 2.2**). The aim of Chapter 3 is to address how these drivers affect the transport, introduction, establishment and spread of invasive alien species.

Drivers do not act in isolation; status and trends in nature are the outcome of the often multiplicative effects of many co-occurring drivers (IPBES, 2019; **Chapter 4**, **Box 4.5**). A recent study systematically and quantitatively ranked direct drivers of change in nature (**section 3.1.2**) in terms of impacts on biodiversity and found that land-use change was generally the most important, but that relative importance of drivers varied across realms, IPBES regions and with the biodiversity components considered (Jaureguiberry *et al.*, 2022) and with scales (Bonebrake *et al.*, 2019). Invasive alien species are recognized as being a driver of change in nature and at the same time a component of biodiversity. A global meta-analysis focussing explicitly on identifying extrinsic factors related to invasive alien species richness found that that human population density of an area, which can be a proxy for multiple and often co-occurring drivers such as trade, travel and land-use, was highly correlated to the number of introduced alien amphibians, fish, plants and spiders within that area (**section 3.1.2** and **Figure 3.2**).

3.1.2 Scope and organization of the chapter with reference to the IPBES conceptual framework

The IPBES conceptual framework and Global Assessment recognize invasive alien species as one of five anthropogenic "direct drivers" of change in nature along with climate change, land- and sea-use change, pollution and direct exploitation of natural resources (Díaz *et al.*, 2015; IPBES, 2019). According to this framework, direct drivers have direct physical (mechanical, chemical, etc.) and biological (physiological, ecological, behavioural) effects on nature (biodiversity and ecosystems) which again impact nature's contributions to people (including ecosystem goods and services) and more generally good quality of life (**Chapter 1**, **Figure 1.11**; Díaz *et al.*, 2018). The IPBES invasive alien species assessment refers to these drivers as "direct drivers of change in nature". The magnitude of the impact of these direct drivers of change on nature, and in some cases on nature's contributions to people and good quality of life, is also shaped by five "indirect drivers" of change: human demography, economic development, technological change, the strength of national and international governance (**Glossary**) as well as sociocultural factors **(Figure 3.3)**. These drivers are described as indirect because they do not directly impact nature (i.e., biodiversity and ecosystems), but act through one or more direct drivers of change in nature (Nelson *et al.*, 2006). Indirect drivers impact nature by affecting the level (e.g., magnitude), direction (e.g., increase or decrease) or rate (e.g., change over time) of the direct drivers. For example, the impacts of economic growth (an indirect driver of change in nature) affects biodiversity or ecosystems through the effects of one or more direct drivers, such as land-use change or pollution.

The classification of drivers outlined through the IPBES conceptual framework has proven useful for synthesis and cross-assessment referencing (e.g., IPBES, 2019; Pörtner *et al.*, 2021) but requires specific consideration for this assessment because invasive alien species are simultaneously the focus of the IPBES invasive alien species assessment and one of the five anthropogenic direct drivers of change in nature (**Figure 3.3**; **Chapter 1**, **section 1.6.1**). This implies that indirect drivers of change in nature may directly affect invasive alien species. For example, international trade is classified as an indirect driver of change in nature, yet an important consequence of trade is that it increases the number of invasive alien species introductions worldwide (Hulme, 2021b), and it does so by directly facilitating the transport and introduction of invasive alien species **(Figure 3.3)**. The process of biological invasions (including all stages: transport, introduction, establishment and spread of invasive alien species; **Glossary**) is also influenced by the five anthropogenic direct drivers of change in nature, including interactions amongst invasive alien species (e.g., by causing "invasional meltdown", **Glossary**; **section 3.3.5.1**; **Chapter 1**, **section 1.3.4**). All five primary direct anthropogenic drivers are therefore also considered in this assessment.

In addition to the main anthropogenic direct and indirect drivers recognized by the IPBES framework, biological invasions can be further facilitated by natural drivers and in particular natural hazards such as tsunamis, floods, fire and hurricanes. While the involvement of human activities within the biological invasion process is inherent to the definition of alien species (**Chapter 1**, **Figure 1.1**), these natural drivers can play a major role in both the introduction of alien species from one region where they are alien to new regions, and also aid their establishment and further spread within regions where they are already present as alien species. A further driver not directly addressed in previous IPBES assessments is biodiversity loss, but in the case of invasive alien species it can be seen as a driver that facilitates their establishment and spread, as a result of reduced resistance of altered natural ecosystems to invasive alien species (**Chapter 1**, **section 1.4.3**). Thus, the transport, introduction, establishment and spread of invasive alien species can be facilitated by both direct and indirect anthropogenic drivers of change in nature, as well as by natural drivers and by biodiversity loss. These drivers do not act in isolation but may interact with each other in different and complex ways (**section 3.1.5**). In part as a result of these complexities, the knowledge base is both limited and fragmented, and attribution of cause-effect relationships can be challenging (**sections 3.1.5**, **3.6.1**). To acknowledge and cover these complexities and limitations, the assessment is organized as follows:

Section 3.2 assesses the role that five indirect drivers of change in nature play in the different stages (transport, introduction, establishment and spread) of the biological invasion process **(Table 3.1)**. The indirect drivers examined are:

- sociocultural drivers and social values (including norms, traditions, cultural beliefs, desires, perceptions);
- demographic (including human population density, migration, international crises and urbanization);
- economic (such as international trade and travel, externalities and wealth, inequality and poverty);
- science and technology (including research and communication and breeding/genomic technology);
- policies, governance and institutions (note that Chapter 3 focuses on the unintended facilitation of biological invasions by policies, governance and institutions targeting other societal objectives (i.e., "perverse incentives", *sensu* IPBES, 2019). Policies, governance and institutions explicitly dealing with biological invasions are dealt with in **Chapter 6**).

Section 3.3 examines the role of the five direct drivers of change in nature in influencing the distribution and abundance of invasive alien species **(Table 3.1)**:

- land- and sea-use changes (including introductions from the direct use of alien species in terrestrial, aquatic and marine bioproduction systems as well as landscape and seascape fragmentation, disturbance and deterioration);
- direct exploitation of natural resources (such as species harvesting, hydrological resource harvesting and mining);
- pollution (including eutrophication and nutrients, other contaminants, marine debris and solid waste);
- climate change (including changes in temperature and precipitation regimes and extremes, carbon dioxide (CO_2) enrichments in air and water, fire regimes, sea level rise and assisted colonization);
- invasive alien species (through biotic facilitation and biological control; **Glossary**).

As explained above, two additional drivers are also considered in section 3.4:

- natural hazards (such as hurricanes, earthquakes, tsunamis).
- biodiversity loss and ecosystem resilience (notably reduced biotic resistance to invasion; Glossary).

Section 3.5 then examines multiple, additive or interacting effects among drivers, especially among anthropogenic direct drivers. Due to knowledge gaps (**section 3.6.1**) this section does not provide an exhaustive assessment but focuses on four illustrative examples of two- or three-way multi-driver impacts and their consequences for biological invasions.

Each subsection within sections 3.2 to 3.5 first briefly describes the trends (**Glossary**) and status of the driver(s) considered, then assesses the overall effects of this driver on the transport, introduction, establishment and spread of invasive alien species and, where there is information, notes specific effects on particular biomes, taxa and units of analysis. Due to the complexities of relationships among drivers and biological invasions coupled with the limited knowledge available (**sections 3.1.3** to **3.1.5**, **Figure 3.3**), a systematic literature search across all topics relevant to the chapter was not feasible, and targeted but coordinated literature searches were conducted instead (**section 3.6.2** for details).[2] In these searches, knowledge was extracted from a broad range of sources, including published scientific literature and reports, and the searches were augmented with literature from the cross-chapter literature review[3] on Indigenous Peoples and local communities and invasive

2. Data management report available at: https://doi.org/10.5281/zenodo.5529309
3. Data management report available at: https://doi.org/10.5281/zenodo.5760266

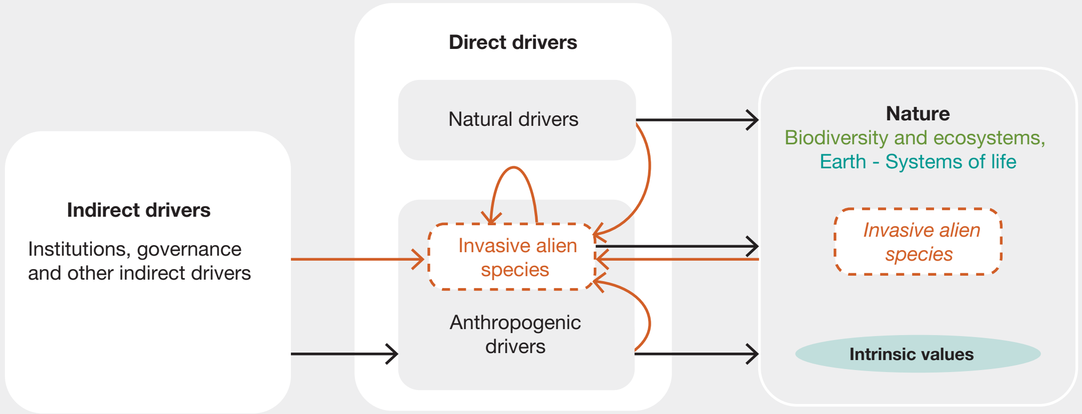


Figure 3 3 **Schematic illustration of how invasive alien species can be influenced by indirect drivers of change in nature, anthropogenic and natural direct drivers of change in nature and by changes to nature (i.e., loss of biodiversity or ecological resilience).**

The assessment builds on the IPBES conceptual framework (Díaz *et al.*, 2015) under license CC BY-NC-SA 3.0, adding focus and detail around how changes to drivers and to nature may influence invasive alien species (lighter arrows). Invasive alien species are classified as one of five direct anthropogenic drivers of change in nature; therefore, they can be influenced both directly and indirectly (*via* other direct drivers) by indirect drivers of change in nature. Biological invasions may also be influenced by all direct drivers of change in nature, including invasive alien species, and by natural drivers (especially natural hazards). Finally, there can also be feedbacks from nature (biodiversity and ecosystems) to invasive alien species, for example, biodiversity loss or invasive alien species can make ecosystems less resistant to further invasive alien species. Invasive alien species are therefore both a driver of change in nature and part of nature (dotted lines). Lighter arrows indicate the focal causal relationships of this chapter, within the context of the IPBES conceptual framework, darker arrows denote additional causal links among drivers and between drivers and nature covered in the IPBES Global Assessment (IPBES, 2019).

alien species. Finally, the authors identify knowledge gaps (**section 3.6.1**) before drawing conclusions and integrating the chapter's findings on the role of drivers of change in nature in facilitating invasive alien species across biomes and realms and across the biological invasion process (**sections 3.6.2**, **3.6.3**).

3.1.3 Identifying drivers of change in nature of relevance for invasive alien species

Chapter 3 adapts the IPBES conceptual framework, recognizing that invasive alien species are, simultaneously, one of the five main direct drivers identified by IPBES (Díaz *et al.*, 2015, 2018; IPBES, 2019), and at the same time the focus of this assessment **(Figure 3.3; Table 3.1)**. Specifically, this chapter acknowledges that some indirect drivers of change in nature, and notably those related to trade, transport and travel, may in fact be directly facilitating the transport, introduction, establishment and spread of invasive alien species. Further, natural drivers, while included in the IPBES conceptual framework as a direct driver, are generally not considered in IPBES assessments (Pereira *et al.*, 2010; Díaz *et al.*, 2015, 2018; IPBES, 2019) but are known to be important factors facilitating biological invasions and are, therefore, included in this chapter (**Figure 3.3**, **section 3.4.1**). Similarly, biodiversity loss can reduce the resilience of ecosystems to invasive alien species and while not considered in the IPBES conceptual framework as a driver, the role of biodiversity loss and changes to ecosystem resilience in facilitating biological invasions is included in Chapter 3 (**section 3.4.2**). Interactions between indirect and direct drivers of change in nature, along with natural drivers and biodiversity loss, create different chains of relationships, attribution, and influences on the biological invasion process. These relationships may vary according to type, intensity, duration and distance. These relationships are captured by cross-referencing between subsections throughout the assessment, and by explicitly considering selected interactive effects (**section 3.5**).

3.1.3.1 Indirect drivers of change in nature affecting invasive alien species

Following previous IPBES assessments (IPBES, 2016a, 2018f, 2018e, 2018c, 2018d, 2019, 2022a), Chapter 3 considers a number of drivers under all five classes of indirect drivers of change in nature: sociocultural, demographic, economic, science and technological and institutional **(Table 3.1)**. It is important to note that indirect drivers of change in nature may both directly and indirectly influence the biological invasion process and the introduction, establishment and spread of invasive alien species **(Box 3.1)**. Sociocultural context, particularly values, beliefs and social norms, can exert significant pressure on decision-making regarding biological invasions (Shackleton *et al.*, 2019; **Chapter 1**, **section 1.5.2**) and is materially manifested in lifestyles and consumption patterns that can act directly in facilitating the introduction and spread of invasive alien species. Demographic drivers, including human population growth and migration, underpin all anthropogenic direct drivers of change in nature that also facilitate biological invasions through increasing urbanization. A significant economic driver that often correlates strongly with the number of alien species found in a country is economic

Table 3 1 **The indirect and direct anthropogenic drivers of change in nature and other factors affecting invasive alien species, as assessed in Chapter 3.**

The IPBES conceptual framework considers indirect and direct drivers of change in nature (Díaz *et al.*, 2015; Nelson *et al.*, 2006). Following the IPBES Global Assessment, this assessment considers five classes of indirect and five classes of direct anthropogenic drivers. In addition, the assessment considers two other classes of drivers: natural drivers and biodiversity loss. For each of these classes of drivers, the assessment considers the influence of a number of specific drivers on the transport, introduction, establishment and spread of invasive alien species. This table shows all the drivers considered in this chapter, with classes of drivers in bold, and drivers under each class in normal font (see **section 3.1.2** for more details).

INDIRECT DRIVERS Anthropogenic factors that affect nature indirectly by altering one or more direct drivers, but which may act both indirectly and directly on invasive alien species.	**ANTHROPOGENIC DIRECT DRIVERS** Factors that describe direct human influence on nature. These may affect invasive alien species directly, or *via* interactions and feedbacks involving other drivers.	**NATURAL DRIVERS AND BIODIVERSITY LOSS** Factors that describe natural drivers and aspects of biodiversity loss which may directly and in interaction with other drivers facilitate invasive alien species.
Sociocultural drivers and social values (3.2.1) **Demographic drivers (3.2.2)** • Regional and national changes in human population density • Human migration • International crises: armed conflict and emergency aid • Urbanization **Economic drivers (3.2.3)** • International trade and global commerce • Human international travel for commerce and tourism • Externalities of negative impacts and cost • Wealth, inequality and poverty **Science and Technology (3.2.4)** • Research • Development of communication technology • Breeding and genomic technologies **Policies, governance and institutions (3.2.5)**	**Land- and sea-use change (3.3.1)** • Introductions from the use of alien species in terrestrial, aquatic and marine bioproduction • Fragmentation of ecosystems • Creation of anthropogenic corridors • Deployment of marine infrastructure • Changes in landscape – seascape disturbance regimes • Landscape – seascape degradation **Direct exploitation of natural resources (3.3.2)** • Species harvesting • Hydrological resources • Fossil fuels and mining **Pollution (3.3.3)** • Eutrophication and nutrient deposition • Other contaminants in water and soil • Marine debris • Dispersal of solid waste **Climate Change (3.3.4)** • Temperature change • Precipitation change • Climate extremes • CO_2 enrichment in air, water • Fire regime changes • Sea level rise • Assisted colonization **Invasive alien species (3.3.5)** • Biotic facilitation • Biological control	**Natural drivers (3.4.1)** Natural hazards such as tsunamis, hurricanes, earthquakes, wildfire, floods and volcanic activity **Biodiversity loss and ecosystem resilience (3.4.2)**

growth, often expressed as per capita gross domestic product (GDP; Dawson *et al.*, 2017; Essl *et al.*, 2011, 2015; Hulme, 2011b), reflecting the intensity of international trade which is a major conduit for the introduction of alien species (Hulme, 2009, 2021b; Seebens *et al.*, 2015; Westphal *et al.*, 2008). While technology is a major factor in economic growth, its effect on biological invasions depends on how it is used. The use of new technologies to limit the transport, introduction, establishment and spread of invasive alien species, along with technological approaches to aid eradication and containment (**Glossary**) of invasive alien species are discussed in **Chapters 5** (**sections 5.5.3**, **5.5.4**) and **6** (**sections 6.3.3.4**, **6.7.2**). **Chapter 3** focuses on the role of technology as a driver, for example, how internet commerce (Walters *et al.*, 2006) is facilitating the introduction and spread of invasive alien species, and on how new technologies such as gene editing can potentially be used to breed species with traits that might make them more likely to be invasive (e.g., cold tolerance, pest resistance). Economic drivers are strongly linked to institutional drivers, which govern production through regulations, taxes and subsidies. The role of policies and institutions in managing biological invasions is addressed in **Chapter 6** whereas the synthesis in **Chapter 3** is restricted to how regulations, taxes and subsidies result in unintended consequences that facilitate the transport, introduction, establishment and spread of invasive alien species (**section 3.2.5**).

3.1.3.2 Anthropogenic direct drivers of change in nature affecting invasive alien species

In line with previous IPBES assessments (IPBES, 2016a, 2018f, 2018e, 2018c, 2018d, 2019, 2022a), this assessment considers the five main anthropogenic direct drivers of change in nature: land-use (including sea-use) change, direct exploitation of natural resources, pollution, invasive alien species and climate change. Land- or sea-use changes can lead to the increased introduction of alien species, either intentionally through the specific use of alien crops and livestock or unintentionally as contaminants of agricultural or aquacultural commodities. Land-use change that leads to habitat (**Glossary**) fragmentation, establishes infrastructure corridors (e.g., roads, canals) through which alien species can spread, alters the baseline rates of disturbance, or more generally degraded habitats can increase the vulnerability of native ecosystems to invasive alien species (Vilà & Ibáñez, 2011). Direct exploitation of natural resources includes both the direct exploitation of biotic resources through species harvesting as well as of abiotic resources such as water and minerals. Harvesting of top predators can lead to trophic cascades that facilitate the establishment of alien species, such as the case of overfishing in the Black Sea resulting in an outbreak of *Mnemiopsis leidyi* (sea walnut; Daskalov *et al.*, 2007). Exploitation of abiotic (e.g., mining) and biotic (e.g., deforestation) resources can drive biological invasions by altering the baseline disturbance regime, which can facilitate the invasion of alien species that are better adapted to the altered conditions (Catford *et al.*, 2012). Pollution, particularly eutrophication, can favour alien species in both aquatic (Vermonden *et al.*, 2010) and terrestrial ecosystems (Brooks, 2003). Climate change, particularly through the effects of higher temperatures and frequency of extreme events, has long been widely expected to increase the rate at which alien species are introduced, establish and spread (Walther *et al.*, 2009). Less well understood is the risk that deliberate translocation of species by humans from one region to another in order to ensure survival in the face of climate change might result in the introduction of invasive alien species (Ricciardi & Simberloff, 2009). Although including invasive alien species as a direct driver affecting biological invasions might sound like circular reasoning, there is increasing evidence of the role that invasive alien species play in facilitating other alien species at different stages of the biological invasion process, aiding dispersal and transportation or as mutualists (e.g., alien mycorrhiza, pollinators and seed dispersers), allowing their reproduction and spread. This process, by which a group of alien species facilitate one another, increasing the likelihood of survival and/or of ecological impact (Braga, Gómez Aparicio, *et al.*, 2018) and potentially causing "an accelerating accumulation of introduced species", has garnered its own term "invasional meltdown" (Simberloff, 2006, **Chapter 1**, **section 1.3.4**; **Chapter 4**, **section 4.7.2**).

3.1.3.3 Natural drivers and biodiversity loss as direct drivers affecting biological invasions

Changes in biodiversity and ecosystems due to natural drivers (including natural hazards) are viewed as innate and integral processes and components of nature itself, and have thus not been extensively considered in prior IPBES assessments (e.g., IPBES, 2019). However, natural drivers can play a significant direct role in the transport, introduction, establishment and spread of invasive alien species both within and beyond their prior invaded range (**Glossary**). Natural large-scale disturbances, such as hurricanes, earthquakes and tsunamis can facilitate the further introduction of alien species from an existing invaded range to new regions, consequently expanding the invaded range (Carlton *et al.*, 2017) as well as facilitating their wider spread in regions where they are already present as alien species (Bellingham *et al.*, 2005). For example, hurricanes appear responsible for the spread of *Cactoblastis cactorum* (cactus moth) between Caribbean islands (Andraca-Gómez *et al.*, 2015), the expansion of *Phragmites australis* (common reed) in the Gulf of Mexico (Bhattarai & Cronin, 2014), and increased rates of recruitment and persistence of invasive alien trees in the subtropical forests of Puerto Rico (Thompson *et al.*, 2007). While native biodiversity is a major component of nature and nature's contributions to people, and thus has been considered as a key response variable in previous IPBES assessments (IPBES, 2019), for invasive alien species biodiversity loss can also be seen as a driver that facilitates biological invasions since loss of biodiversity, and especially reduced functional complexity and/or integrity of ecosystems, can reduce biotic resistance to invasive alien species (Levine *et al.*, 2004). Chapter 3 thus considers the consequences of native biodiversity loss and changes to ecosystem resilience for biological invasions, since this is known as an important feedback mechanism directly influencing the introduction, spread and establishment of invasive alien species (**Figure 3.3**; **Table 3.1**; **section 3.4.2**). This additional complexity is explicitly captured by assessing focal relationships (lighter arrows) from natural drivers and from nature to invasive alien species, in the context of the IPBES conceptual framework **(Figure 3.3)**.

3.1.4 Differential role of drivers along the stages of the biological invasion process

Biological invasions are widely viewed as processes comprising a series of sequential stages (Blackburn *et al.*, 2011; Colautti & MacIsaac, 2004; Theoharides & Dukes, 2007; **Chapter 1**, **section 1.4**). These stages capture the transport of a species to a region beyond its native range, the introduction of the species (intentionally or/ and unintentionally) into habitats in that region, and its subsequent establishment as a self-sustaining population, followed by its wider geographic spread in the invaded range (**Figure 3.4**; **Chapter 1**, **section 1.4**). Pathways of introduction are referred to by the Convention on Biological Diversity as the means by which species are moved to new regions beyond their native range (CBD, 2014; Hulme *et al.*, 2008). Pathway assessment usually focuses on movements until a species reaches the border of an administrative unit, such as a country, although is not restricted to this definition. Pathways are categorized into six major classes (release, escape, contaminant, stowaway, corridor, and unaided) with several sub-classes (CBD, 2014; **Chapter 2, Table 2.1**; **Chapter 1**, **Box 1.6**). In the transport stage, drivers can act by facilitating pathways, such as when economic growth increases trade and transport volumes, thereby facilitating transport of alien species as stowaways (**section 3.2.3**). In later stages of the biological invasion process, drivers can act both intentionally and unintentionally to facilitate the establishment and spread of invasive alien species, such as when alien species are used for or spread as pests or contaminants of goods used for bioeconomic purposes (agriculture, aquaculture, forestry, or as pets or ornamentals)

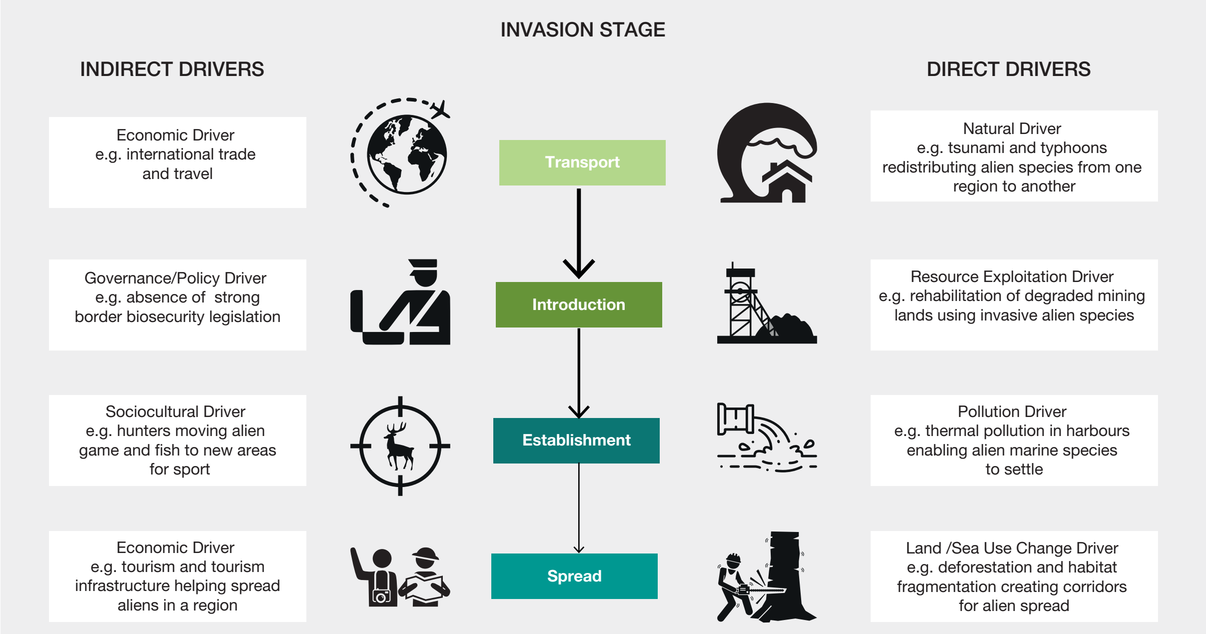


Figure 3 4 **Schematic using selected examples of how both indirect and direct drivers of change in nature may facilitate invasive alien species along four stages in the biological invasion process: transport, introduction, establishment and spread.**

The examples are meant to be illustrative and not an exhaustive set of scenarios with several drivers mentioned also influencing other stages. See **section 3.6.2** for a synthesis of the influence of different drivers on invasive alien species across stages of the biological invasion process, biomes and realms.

or when land-use change or biodiversity loss causes natural ecosystems to be less resistant to biological invasions (**sections 3.3.1**, **3.4.2**).

The biological invasion process is central to the quantitative risk assessment of invasive alien species (Leung *et al.*, 2012; **Chapter 5**, **Figure 5.1**). Nevertheless, while considerable amounts of data are being captured on the dynamics of invasive alien species across some of these biological invasion stages (Abellan *et al.*, 2016; Essl *et al.*, 2015; Gravuer *et al.*, 2008; Moodley *et al.*, 2013; Renault *et al.*, 2018; F. Ribeiro *et al.*, 2008), a full understanding of the causal factors responsible for successful introductions of invasive species remains more limited (Puth & Post, 2005). Species' traits that facilitate the introduction, establishment and spread of particular taxa (McGregor *et al.*, 2012; Moodley *et al.*, 2013; Ribeiro *et al.*, 2008) or direct drivers, such as climate change (Hulme, 2017), have been the focus of the current understanding of the transitions among different biological invasion stages.

Therefore, Chapter 3 aims to systematically[4] examine how different direct and indirect drivers of change in nature, as well as natural drivers and biodiversity loss, may influence each of the four biological invasion stages: transport, introduction, establishment and spread **(Figure 3.4)**. Specifically, each section examines the evidence for each driver within the context of the different biological invasion stages across terrestrial and aquatic biomes and for the major realms or taxonomic groups (plants, invertebrates, vertebrates and microbes; see **sections 3.1.3** and **3.6.1** for details on the search strategy and the knowledge base extracted, respectively).

4. Data management report available at https://doi.org/10.5281/zenodo.5529309

3.1.5 Attributing causality and understanding interactions among drivers

A number of studies suggest recent increases in numbers of alien species (**Chapter 2**; Seebens *et al.*, 2017) are likely augmented by increases in the rates of movement of goods and people (Essl *et al.*, 2019; Murphy & Cheesman, 2006). Due to lagged responses (time lags; or lag phase in the **Glossary**), especially towards the later stages of the biological invasion process, consequences of recent increases in transport and travel are unlikely to be fully realized at present, resulting in potentially quite substantial

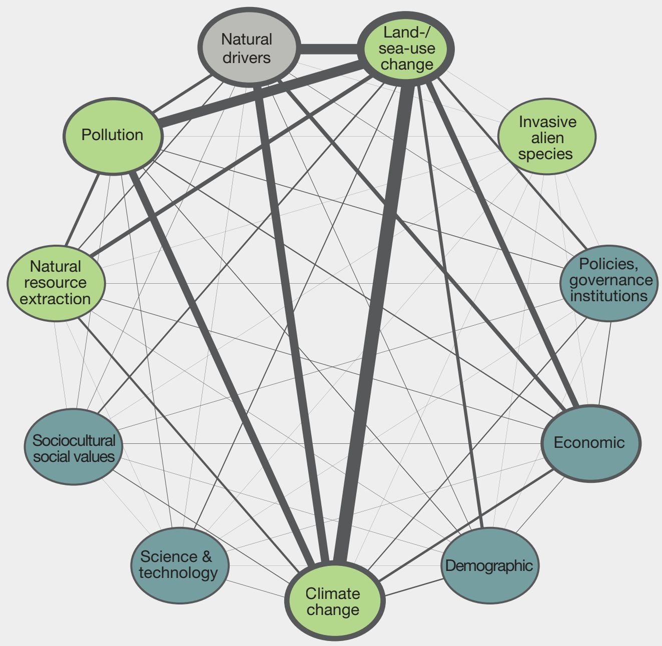


Figure 3 5 **Network diagram illustrating the extent of knowledge on indirect (blue circles), direct (green circles) anthropogenic drivers of change in nature and natural drivers (grey circle), and their interactions (lines).**

The thickness of the lines between drivers is indicative of the number of papers that jointly addressed the two linked drivers. The size of the line surrounding each circle reflects the number of papers listed by Web of Science between 2000 and 2019 generated by a topic search on a particular driver in relation to invasive alien species. Note the greater emphasis on direct drivers both individually and jointly. Note that effects of biodiversity loss on biological invasions are not included in this figure. A data management report for this figure is available at https://doi.org/10.5281/zenodo.7861123 "invasion debts" (**Glossary**; Essl *et al.*, 2011; Rouget *et al.*, 2016; **Chapter 1**, **section 1.4.4**; **Chapter 2**, **section 2.2.1**). As a consequence of increasing rates of introduction, possibly aggravated by climate change, many historical and contemporary invasive alien species are now increasingly emerging as threats to modern agriculture and food security (Subbarao *et al.*, 2015; **Chapter 4**, **sections 4.4**, **4.5**, **4.6.2** and **4.6.3**). However, attributing such global patterns of increasing rates and impacts of invasive alien species to specific drivers, such as travel, trade, or migration, is difficult.

The drivers that directly or indirectly facilitate biological invasions are correlated and causally linked through a series of co-occurring global change trends (**section 3.1.1**). The different stages of the biological invasion process can be affected by different sets of drivers (**section 3.1.4**), and drivers can interact in complex ways to facilitate the biological invasion process (**section 3.5**). While indirect drivers of change in nature may act directly on biological invasions, both natural direct drivers and anthropogenic direct drivers of change in nature can also have indirect effects on biological invasions through their influence on other drivers or *via* feedbacks from biodiversity loss **(Figure 3.3)**. For example, climate change (direct driver of change in nature) can have a direct driving effect on land-use change e.g., through a shift to more intensive agriculture which could lead to a direct effect on biodiversity loss and thus facilitate the introduction of invasive alien species. Similarly, expanding urbanization (an indirect driver ofchange in nature) can lead to increased exploitation of hydrological resources, increased pollution, as well as habitat fragmentation (direct drivers of change in nature). All these factors may increase the extent of invasive alien species, either alone or in concert, and when in concert they may act additively or multiplicatively. Unfortunately, this complexity is rarely captured in studies of invasive alien species and research often attempts to address a single proximate cause of biological invasions rather than teasing apart multiple factors or disentangling the chain of causation from indirect to direct drivers (Hulme, 2022). The multi-driver studies that exist tend to focus on interactive effects of a few key direct drivers of change in nature, notably land-use change, pollution and climate change **(Figure 3.5)**. As a result of these complexities, the knowledge base is both limited and fragmented, and attribution of cause-effect relationships can be challenging. See **section 3.1.2** for an outline of how this challenge was tackled across the chapter, **section 3.6.2** for an overview of the resulting evidence-base, and individual driver subsections (**sections 3.2**, **3.3**, **3.4**, **3.5**) for how this was tackled or each driver.

3.2 THE ROLE OF INDIRECT DRIVERS OF CHANGE IN NATURE ON INVASIVE ALIEN SPECIES

Following previous IPBES assessments (IPBES, 2018a, 2019), the five classes of indirect drivers examined in this chapter are sociocultural drivers and social values (**section 3.2.1**), demographic drivers (**section 3.2.2**), economic drivers (**section 3.2.3**), science and technology drivers (**section 3.2.4**), and finally policies, governance and institutions drivers (**section 3.2.5**). Invasive alien species are classified as a direct driver of change in nature in the IPBES scheme (Díaz *et al.*, 2015) which implies that indirect drivers of change in nature can directly influence invasive alien species (**Figure 3.3**; **sections 3.1.3** and **3.1.4**). The specific mechanism of how these influences occur is discussed under each driver below.

3.2.1 Sociocultural drivers and social values

Sociocultural contexts, particularly social values which are created by social norms, traditions, cultural beliefs, and accepted morally by society, can exert significant pressure on decision-making regarding invasive alien species (Shackleton *et al.*, 2019; **Chapter 1**, **sections 1.5.2** and **1.5.3**). Additionally, sociocultural drivers and social values are manifested in lifestyles and consumption patterns, which act as indirect and direct drivers of change in nature and affect the transport, introduction, establishment and spread of invasive alien species **(Figure 3.6)**. Sociocultural drivers thus interact with other indirect drivers, especially demographic drivers such as changes in population density (**section 3.2.2.1**), migration (**section 3.2.2.2**) and urbanization (**section 3.2.2.4**), as people have a long history of exchanging new species and bringing them with them for ornamental, cultural and practical use. Sociocultural drivers further interact with economic drivers, such as trade (**sections 3.2.3.1**, **3.2.3.2** and **3.2.3.3**) and travel (**section 3.2.3.4**) as well as science and technology drivers, such as communication technology (**section 3.2.4.2**). Sociocultural drivers can influence the rate and magnitude of change in a number of direct drivers of change in nature, particularly related to land- and sea-use changes (**section 3.3.1**), but also species harvesting (**section 3.3.2.1**), pollution (**section 3.3.3**) and drivers related to biodiversity and ecosystem health such as unintended consequences of the intentional introduction of invasive alien species (**section 3.3.5.2**) and biodiversity loss (**section 3.4.2**).

Some alien species are associated with cultural, aesthetic or practical value (**Chapter 1**, **section 1.5.2** and **Chapter 4**,

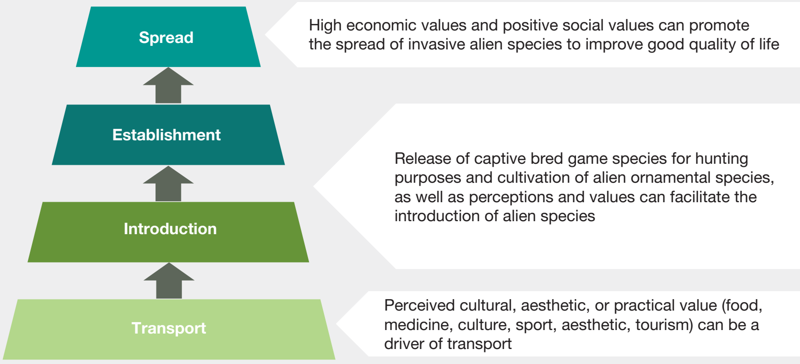


Figure 3 6 **Examples of the role of sociocultural drivers and social values in facilitating invasive alien species across stages of the biological invasion process.**

Illustration of how sociocultural drivers and social values may interact with or operate through a variety of other direct and indirect drivers of change in nature, including economic drivers, harvesting of natural resources and land- and sea-use change, to facilitate invasive alien species across the stages of the biological invasion process.

sections 4.5 and **4.6.3**), and such values can indirectly act to facilitate the introduction of these alien species. Indeed, plants that are economically valuable were shown to be 18 times more likely to become naturalized than those that are not (van Kleunen *et al.*, 2020). In Aboriginal societies in Australia, not only plants important as food and materials but also species of ritual and cultural importance have been intentionally translocated and grown for their societal and cultural benefits (Silcock, 2018). The replacement of native crayfish in Spain with two alien species intentionally introduced from North America to satisfy local tastes is a good example of the economic and social value of promoting the substitution of native species (**Glossary**) with invasive alien species (Clavero, 2016). In contrast, awareness of the adverse impacts of invasive alien species on nature, nature's contributions to people and good quality of life can lead to an increase in action by people and consequently be a driving force behind preventing the introduction of invasive alien species (McNeely, 2001; Shackleton *et al.*, 2019).

The decision to intentionally introduce an alien species or not is largely dependent on the balance between the perceived benefits of specific alien species and the perceived costs of adverse impacts. Therefore, social values have considerable influence on the judgment of whether or not to introduce an alien species (**Chapter 4**, **Box 4.2**). Estévez *et al.* (2015) reported that conflicts over invasive alien species arose primarily from differences in value systems (utilitarian, moralistic, humanistic, negativistic), rather than differences in benefit or risk perceptions between different stakeholder groups and decision-makers (**Chapter 1**, **section 1.5.2**; **Chapter 5**, **section 5.6.1.2**). According to this study, salmonids in South America, alien species of *Acacia* spp. in Africa and *Dreissena polymorpha* (zebra mussel) in Europe are examples of utilitarian vs. naturalistic value-system conflicts whereas alien mammals (example in **Box 3.2**) and trees have caused moralistic or humanistic vs. naturalistic or negativistic value conflicts in all IPBES regions (Estévez *et al.*, 2015). In another global study, Kapitza *et al.* (2019) found that the local public was more likely to focus on sociocultural benefits whereas academics focused on nativeness, and stakeholders from Africa were more likely to identify ecological benefits whereas Europeans were less likely to identify ecological and sociocultural benefits. Contextual factors, such as stakeholder role, socioeconomic status, time since the introduction occurred, and region therefore also affect the overall valuation, which again can impact behavioural choices or actions influencing biological invasions (Kapitza *et al.*, 2019; Shackleton *et al.*, 2007).

There have been many intentional introductions of invasive alien species, motivated by desires to improve specific aspects of good quality of life (**Box 3.2** and **Chapter 2**, **section 2.1.2**). Though the perception that an invasive alien species confers a benefit is not necessarily knowledge-based, it nonetheless serves as a powerful motivation for the introduction of invasive alien species usually as a result of expectations relating to increased employment, wealth, food sources, or other material gains (**Chapter 4**, **section 4.1.2**; **Chapter 5**, **section 5.6.1.2**). Meanwhile, negative impacts related to invasive alien species (whether they are intentionally or unintentionally introduced) such as threats to good quality of life, often motivates the management of biological invasions and specifically the control of invasive alien species (McNeely, 2001; Shackleton *et al.*, 2019). However, different perceptions can and often do co-exist. Specific invasive alien species considered as problematic by one social sector may provide valuable

Box 3 2 The role of hunters intentionally spreading game animals.

Introduction for hunting accounts for a large proportion of intentional introductions of invasive alien birds and mammals, both in absolute numbers and compared to introductions for biological control, pet trade and use of fur (Carpio *et al.*, 2020, 2017; Genovesi *et al.*, 2009; Hulme *et al.*, 2008). Carpio *et al.* (2017) found stocking for hunting to be a dominant source of introductions of invasive alien species. High rates of introduction and establishment can result from intensive human efforts to maintain sufficiently large and stable populations of alien species for hunting (Champagnon *et al.*, 2012, 2016). Introductions may also occur in cases where population sizes of traditionally-used native species have significantly reduced, and alien species are then introduced to supplement hunting or fishing (Carpio *et al.*, 2017; Clavero, 2016), or when alien species are introduced for the diversification of species available for hunting (Carpio *et al.*, 2017). Such efforts and considerations have resulted in large-scale introductions of a number of alien birds and mammals as game species throughout Europe, generating significant revenues through licensing fees and through creating a demand for hunting gear and services. As societal and cultural views on game species vary from being a valuable food resource *via* recreational activity to being an ecological nuisance, policy can follow suit (Duffy & Lepczyk, 2021). Alien species are still released for hunting purposes in Europe, but the rate of new species introduced has been declining over the past 10 years as the knowledge that alien species have negative effects on native ecosystems has increased (Carpio *et al.*, 2017). This increase in knowledge may have contributed to reducing the number of alien species introductions in recent decades (Fèvre *et al.*, 2006). In addition, game managers have criticized the use of alien species for hunting from an ecological perspective (Delibes-Mateos, 2015), and hunters indicate that they favour hunting wild game in biodiversity-rich landscapes rather than released individuals, and are willing to pay at least 20 times more per wild partridge *(Alectoris rufa)* hunted relative to a farm-reared bird (Delibes-Mateos *et al.*, 2014). However, hunters do not always recognize that a game species is alien (Cerri *et al.*, 2016). Furthermore, fines for the illegal importation and/or release of alien species are relatively low and the detection rate of illegal importation is low (Caudell *et al.*, 2016) compared to the economic benefit for hunters and landowners indicating that the introduction of alien species for hunting is still difficult to manage (**Chapter 5**, **Box 5.6**).

Figure 3 7 ***Phasianus colchicus* (common pheasant) killed by a recreational shoot.**

Phasianus colchicus are native to Asia and parts of Europe (Balkans and northern Caucasus) and have been introduced as game birds throughout the world, including Europe, North America, Hawaii, Japan, Australia and New Zealand. In the United Kingdom, pheasants were introduced in the eleventh century and became a popular game bird in the nineteenth century, being widely bred and released for recreational use. The current breeding population in the United Kingdom has 2.3 million female birds (RSPB, 2021). Photo credit: MykolaMoriev, Shutterstock – Copyright.

contributions to people, cultural benefits, or other intrinsic values to another (**Chapter 1**, **section 1.5.2**; McNeely, 2001; Schlaepfer *et al.*, 2011). For example, in poor rural communities in Madagascar, the introduced *Procambarus virginalis* (Marmorkrebs), while acknowledged to be detrimental to rice farming and fishing, is also valued as a cheap and widely accessible protein source for food and feed, and the perception of overall benefit relative to costs is considered high by people not directly involved in fishing or farming and by communities with a long history of crayfish invasion (Andriantsoa *et al.*, 2020). *Opuntia ficus-indica* (prickly pear), an invasive alien cactus, has high economic value to rural communities in South Africa, where it is used for stock fences or as fresh fruit, and as such is considered positively by farmers, which has consequently affected its establishment and spread (Kapitza *et al.*, 2019; Shackleton *et al.*, 2007). Other examples include invasive alien species introduced as game animals **(Box 3.2)** or for recreational fishing.

3.2.2 Demographic drivers

Demographic drivers, including human population growth and movement, are fundamental factors behind many drivers that directly facilitate biological invasions. This section summarizes the evidence for the influence of four main demographic drivers of change in nature: 1) changes in human population density; 2) human migration; 3) international crises, such as armed conflict and emergency relief; and 4) urbanization **(Figure 3.8)**.

Demographic drivers of change in nature interact with other indirect drivers, especially economic drivers, such as trade (**sections 3.2.3.1** to **3.2.3.3**) and travel (**section 3.2.3.4**) and science and technology drivers, such as breeding and genomic technologies (**section 3.2.4.3**). Demographic drivers operate by influencing the rate and magnitude of change in a number of direct drivers, most obviously those related to land- and sea-use change (**section 3.3.1**), but also other direct drivers such as species harvesting (**section 3.3.2.1**), water extraction (**section 3.3.2.2**), pollution (**section 3.3.3**), climate change (**section 3.3.4**) and drivers related to biodiversity and ecosystem health such as unintended consequences of the intentional introduction of invasive alien species (**section 3.3.5.2**) and biodiversity loss (**section 3.4.2**).

3.2.2.1 Regional and national changes in human population density

The world's population has doubled over the last 50 years (IPBES, 2019) and is expected to reach 8.5 billion people in 2030, of which approximately 60 per cent (about 5 billion) will reside in urban areas (**section 3.2.2.4**; United Nations *et al.*, 2019). Coastal areas are experiencing faster growth rates and 51 per cent of the world's population will live within 100km of the coast by 2030 (Kummu *et al.*, 2016). Approximately 75 per cent of the two billion people to be added to the global population by 2050 will live in sub-Saharan Africa (about 50 per cent) and Central and Southern Asia (about 25 per cent) (United Nations *et al.*, 2019). Regions with high human population densities are often associated with high rates of species introductions and establishment of alien species (Pyšek *et al.*, 2020), and have been associated with both intentional and unintentional transport of species to locations outside of their native ranges (Hulme, 2009; Levine & D'Antonio, 2003). Human population density growth enhances regional trade (**section 3.2.3.1**; United Nations *et al.*, 2019), intensifies

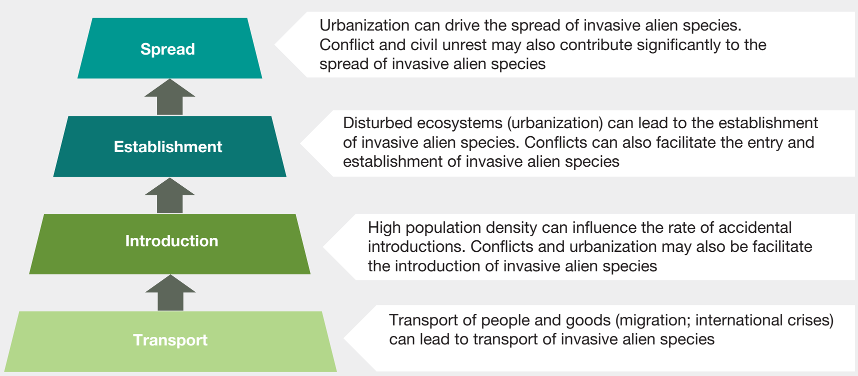


Figure 3 8 **Examples of roles of demographic drivers in facilitating invasive alien species across stages of the biological invasion process.**

Illustration of how demographic factors such as urbanization, international crises and movement of people, directly or *via* other drivers such as land-use change, can facilitate invasive alien species across the stages of the biological invasion process.

urbanization (**section 3.2.2.4**), and increases pressures from a suite of land-use related factors, including nature-based industries (**section 3.3.1.1**), and is associated with the loss, fragmentation and degradation of natural ecosystems (**sections 3.3.1.2** to **3.3.1.5**; IPBES, 2019). These are all factors known to promote alien species in terrestrial (Pyšek *et al.*, 2020) and marine environments (C. C. Murray *et al.*, 2014) across multiple taxonomic groups (Bellard *et al.*, 2016). Evidence from several regional and global studies (Dawson *et al.*, 2017; Essl *et al.*, 2019) shows that alien species richness is positively associated with human population density with human population density-related processes acting on different stages of the biological invasion process (Pyšek *et al.*, 2010). In a global study of a wide taxonomic range of established alien species (**Glossary**), human population density was shown to have the strongest influence on fish, plants and spiders whereas weaker but positive relationships were found for ants, birds, mammals and reptiles (Dawson *et al.*, 2017). In Europe, human population density is positively associated with increased alien species richness for a wide range of plant and animal groups, with highest alien species richness values occurring in regions with more than 91.1 inhabitants/km^2 and the lowest values in regions with fewer than 8.5 inhabitants/km^2 (Pyšek *et al.*, 2010). At the national scale, a comprehensive national alien species assessment in Norway, covering all multicellular organisms, found that for all taxonomic groups considered (terrestrial and aquatic plants, invertebrates, fungi and vertebrates) alien species richness was positively correlated with human population density (Sandvik *et al.*, 2019). Similarly in the United Kingdom, human population density was a good predictor of freshwater fish introductions (Copp *et al.*, 2010). In Brazil, Fonseca *et al.* (2019) showed that human population density influenced the rate of unintentional introductions, resulting in high records of alien amphibians and reptiles in densely-populated areas. At more local scales, evidence shows that increased human population density in surrounding areas is a significant predictor of alien species richness for plants within national parks in the United Stated of America (McKinney, 2002) and for multiple alien taxa (including birds, mammals, vertebrates and plants) in South Africa's Kruger National Park (Spear *et al.*, 2013). The number of visitors (McKinney, 2002) or occupants (Foxcroft *et al.*, 2008) within a national park is positively correlated with alien species richness. A case study on urban wetlands further highlights that alien species richness is correlated with human population density, however found that alien herbs, shrubs and trees all respond differently to human pressures (Ehrenfeld, 2008).

Human population density can be used as a proxy to better understand the role of various human activities across stages of the biological invasion process, and has been shown to facilitate both introduction (Gallardo & Aldridge, 2013; Pyšek *et al.*, 2010) and establishment (Dawson *et al.*, 2017). This may be related to human population density acting as a proxy for propagule pressure (**Glossary**) and the intensity of anthropogenic disturbance (**section 3.3.1**), two mechanisms known to facilitate the introduction and establishment of alien species, respectively (Gallardo & Aldridge, 2013; Roura-Pascual *et al.*, 2011). For example, in Europe, the introduction of alien mammals was significantly correlated with human population density whereas establishment success was not (Jeschke & Genovesi, 2011). At the global scale, other socio-economic factors, such as per capita GDP and proportion of agricultural land, appear to be more important predictors of relative invasive alien species richness than population density across both islands and mainland regions (Essl *et al.*, 2019; Westphal *et al.*, 2008). Essl *et al.* (2019) show that human population density had a greater influence on absolute alien species richness in mainland regions compared to islands, and this pattern was more pertinent for established alien species than for the subset of alien species that were invasive. This illustrates that while population density is correlated with invasive alien species dynamics, the relationship is complex so that countries that have high rates of population growth are not necessarily those with high rates of introduction of invasive alien species.

The review of the literature for this chapter (**section 3.6.1**)[5] highlighted that the majority of the evidence for human population density as a driver that facilitates biological invasions can be found in the terrestrial realm, followed by the freshwater realm, with few marine examples to draw from despite the large concentration of human populations living near to the coast. The IPBES regions of Oceania and Asia-Pacific were the least well studied, with Europe and the Americas having the greatest focus. Despite the other regions (e.g., Africa) being included in global studies, they have relatively few examples to draw from and examples are often from a small selection of countries (e.g., South Africa). Dawson *et al.* (2017) provide a global synthesis of the taxa, however microbes and invertebrates remain poorly studied.

3.2.2.2 Human migration

While the term migrant has specific definitions, Chapter 3 specifically focuses on people moving away from their place of usual residence to take up residence in another country. The rate of migration is increasing: in 2019, 3.5 per cent of the global population (272 million people) were living in a country other than their country of birth, compared with 2.8 per cent in 2000 and 2.4 per cent 1980 (International Organization for Migration, 2019; Vidal *et al.*, 2018). Migrating humans act as direct dispersal vectors (**Glossary**) in the transport and spread of plants, animals and microbes, either unintentionally in the case of pests and diseases or

5. Data management report available at https://doi.org/10.5281/zenodo.5529309

intentionally in the case of pets, livestock, ornamentals, or crops. Sociocultural drivers and social values (**section 3.2.1**) are important factors behind many of these intentional introductions. Human migration thus operates through and in synergy with other indirect drivers of change in nature such as changes in travel, trade and transport (**sections 3.2.3.1** to **3.2.3.4**), urbanization and/or abandonment of land (**section 2.3.3.4** and **3.3.1.5.1**) and population changes, armed conflict and emergency relief (**sections 3.2.2.1** and **3.2.2.3**).

Generally, broad-scale analyses of the spread and distribution of alien species find links between the rate and origins of introductions of new species and human migration history. For example, early introductions of invasive alien plants in Brazil and Australia can be linked to waves of European migration (Phillips *et al.*, 2010; Zenni, 2014), patterns of early bird introductions worldwide spatially and temporally tracked the expansion of European (and especially British) colonialism (Dyer *et al.*, 2017). Patterns of ant invasion dynamics globally bear clear imprints of human demographic patterns including an early wave of biological invasions (1850-1910) coinciding with a period of high human migration (Bertelsmeier *et al.*, 2017). As a more recent example, the number of insect invasions in Europe increased steeply in response to political changes that allowed increased movement of people and goods in Europe after the fall of the Iron Curtain in 1989, and also increased following the expansion and integration of the European Union (Roques *et al.*, 2016). Similarly in line with geopolitical and economic trends, vertebrate introductions from Europe to the United States of America peaked in the nineteenth century paralleling high rates of human migration in the same direction, while alien vertebrate introductions in the opposite direction (United States to Europe) are at their highest now (Jeschke & Strayer, 2005).

Migration can also drive introductions and spread of alien species within regions, for example, immigrants to South Africa from other countries in the region have brought with them their own medicinal plants and have created a market for them (Faulkner *et al.*, 2020). A parallel line of evidence for the role of historic human migration on alien species' distributions comes from intraspecific genetic patterns in alien species. Several studies reveal close congruence with historical large-scale human migration, including for human diseases carried directly by humans or by vectors associated with humans (e.g., Conn *et al.*, 2002), pests (e.g., Puckett & Munshi-South, 2019) and even parasites of pests (e.g., Aketarawong *et al.*, 2015; United Nations Department of Economic and Social Affairs, 2019). Accordingly, political or other barriers to human migration can impede rates of spread of alien species, as is seen in the spread of alien insects across Europe, which appears to have been hampered by the political East-West barrier during the Cold War, as rates of spread were four times higher after 1989 compared to the 1950-1989 period (Roques *et al.*, 2016).

While human migration is increasing globally (United Nations Department of Economic and Social Affairs, Population Division, 2020), other movements of people and goods around the globe resulting from trade and direct use of alien species in nature-based industries (**sections 3.2.3.1**, **3.3.1.1**) and travel and tourism (**section 3.2.3.4**) are increasing even more rapidly (UNWTO, 2021). The relative importance of human migration *per se* in the introduction of invasive alien species is therefore likely to be decreasing. In line with this, recent global analyses reveal only weak influence of human migration on recent patterns in the rate of alien species' introductions, and also weak influences of human migration rates on variation in biological invasions among taxonomic groups and between geographic regions (Seebens *et al.*, 2015). However, human migration is predicted to rise as climate change displaces people from drought-, flood- or storm-hit regions (Rigaud *et al.*, 2018), indicating that human migrations, possibly in interaction with climate change, land-use change or natural drivers, could contribute to increased rates of introduction of invasive alien species in the future. Pressure from invasive alien species could be expected to be highest in countries with the highest inward migration: currently United States, Germany, Saudi Arabia, Russian Federation and the United Kingdom (United Nations Department of Economic and Social Affairs, Population Division, 2020).

While the review of the literature on human migration as a driver facilitating biological invasions undertaken in Chapter 3 is limited in extent, authors considered evidence from plants, vertebrates, invertebrates and microorganisms, and from all IPBES regions, with evidence dating back to colonial times (**Chapter 2**, **section 2.2.1**). As such this section broadly covers relevant variability in terms of geography, taxonomy and realms and time. The literature on migration as a driver facilitating biological invasions is focused on the transport and especially introduction of alien species, with less evidence from the latter stages of the biological invasion process.

3.2.2.3 International crises: armed conflict and emergency relief

International crises, specifically armed conflicts and humanitarian emergency relief and development assistance operations can be powerful indirect drivers of change in nature that may directly facilitate the transport, introduction, establishment and spread of invasive alien species (**Chapter 6**). By 2020, the global number of refugees and asylum seekers was approximately 25 million people (UNHCR, 2020). Both crises and aid can affect biological invasions through abrupt and substantial changes in travel, trade and transport (**sections 3.2.3.1** to **3.2.3.4**), and through substantial and rapid human population movements and changes (**sections 3.2.2.1** and **3.2.2.2**). These trade, transport and population changes affect biological invasions through influencing, broadly, the movement of goods and deployment of infrastructure, and the transport and movement of humans and their luggage, all of which can act as direct dispersal vectors (assisting the uptake and introduction of alien species both intentionally or unintentionally; **section 3.3.1.1**). Crises and aid can also lead to changes in nature through increasing or decreasing the intensity of land- and sea-use (**section 3.3.1.1**), which may again affect biological invasions (e.g., fragmenting landscapes, creating corridors, deploying infrastructure, leading to land-use abandonment, degrading habitats, or changing disturbance regimes; **sections 3.3.1.2** to **3.3.1.6**).

Conflict and civil unrest may contribute significantly to the introduction and spread of invasive alien species through several mechanisms: (i) civil unrest or war can lead to the breakdown of phytosanitary and animal health control and management, and the loss of supply lines for materials as well as to the displacement of large numbers of people; (ii) areas where there is civil unrest or war may be more vulnerable to the introduction of invasive alien species because of the lack of inspections and border controls, facilitating unregulated movement as well as deliberate smuggling of people and their belongings, and also because of the increased movement of military personnel and refugees; (iii) inflows of food aid may also be contaminated with alien species; and (iv) difficulties in obtaining access to border areas because of landmines and other hazards impedes border control (FAO, 2001; Moore, 2005; Murphy & Cheesman, 2006).

War entails transportation of people and goods, deployment of heavy machinery, disturbance of habitats, creation of bare ground, and in some instances intentional introductions of alien species (food supply etc.; Fosberg, 1957). Military transport, equipment and supplies, often covered with dirt or mud from the field, are effective means of dispersal for many alien species (Cofrancesco Jr *et al.*, 2007; Dalsimer, 2002). A number of introductions have been linked to movement of troops and military equipment during the Second World War: *Rattus rattus* (black rat) were introduced to the Midway Islands by navy ships; a desert shrub, *Peganum harmala* (African rue) was introduced inadvertently into New Mexico and Texas *via* airfields; *Tribulus terrestris* (puncture vine) and the agricultural pests *Striga asiatica* (witch weed) and *Globodera rostochiensis* (yellow potato cyst nematode) are believed to have entered North America on returning military equipment; and *Boiga irregularis* (brown tree snake), native to New Guinea, was unintentionally introduced to the island of Guam most likely in military shipments of fruit (Cox, 1999). A root rot of pine trees, *Heterobasidion annosum* (root rot), was inadvertently introduced into Italy by American troops during World War II where it has resulted in an unprecedented mortality rate of *Pinus pinea* (Italian stone pine; Pilcher, 2004). More generally, an increase in marine invasive alien species was observed at Pearl Harbor following World War I and II (Coles *et al.*, 1999). More localized military operations have also led to biological invasions. The introduction of *Diabrotica virgifera* (western corn rootworm) in Serbia in the 1990s was associated with incoming military transport from the United States (EPPO, 1996), and has rapidly become a widespread (**Glossary**) threat to European corn production (Bažok *et al.*, 2021). Habitat disturbance caused by military activity may also facilitate invasive alien species. In Poland, bomb craters were found to have higher numbers of invasive alien plants compared to the surrounding landscape (Krawczyk *et al.*, 2019). Military training activities resulting in soil disturbance facilitated the spread of *Imperata cylindrica* (cogon grass) in military camps in the United States (Yager *et al.*, 2009). Again in the United States, tank traffic activity facilitated invasive alien species in prairie grasslands (Wilson, 1988).

War also facilitates the movement of humans, who may themselves become vectors of alien species including pathogens. For example, the global spread of the Spanish Flu post-World War I was attributed to movement of troops (Neill & Arim, 2011) and a 2010 cholera outbreak in Haiti was attributed to incoming United Nations peacekeeping troops from Nepal (Frerichs *et al.*, 2012). Not all war and crisis-related introductions are unintentional. *Mikania micrantha* (bitter vine) is reported to have been introduced to northeast India to camouflage air strips built by the Allied Forces during World War II to impede the advancing Japanese forces (Kohli *et al.*, 2011; Randerson, 2003). More unique links between armed conflicts and biological invasions also exist, for example, attempts to rectify war damages through replanting activities and making use of invasive alien species in these efforts, i.e., the planting of *Cynodon dactylon* (Bermuda grass) or revegetation of denuded Pacific islands during World War II (Fosberg, 1957). War can also hamper biological invasions, as exemplified by reductions in the rates of global spread of alien ant species due to decreased global trade in both World Wars I and II (Bertelsmeier *et al.*, 2017).

Emergency relief, reconstruction efforts and humanitarian aid after armed conflicts and disasters may also contribute to the introduction and spread of invasive alien species. For example, *Iguana iguana* (iguana), *Osteopilus septentrionalis* (Cuban treefrog) and *Scinax x-signatus* (Venezuela snouted treefrog) were introduced to Dominica, West Indies, *via* emergency relief shipping containers (van den Burg *et al.*, 2019). *Parthenium hysterophorus* (parthenium weed), originally from Mesoamerica, has become invasive in India, where its seeds arrived in grain shipments. It was then spread to Sri Lanka *via* peacekeeping efforts (Kohli *et al.*, 2006; Pallewatta *et al.*, 2003). Aid shipments also resulted in its introduction to Ethiopia (Wittenberg & Cock, 2003).

Such cases have led to local concern when receiving humanitarian aid shipments. For example, due to the highly invasive nature of *Sorghum halepense* (Johnson grass), American Samoa refused the offer of assistance from the Australian International Assistance Program (Tuinoula, 2003).

As illustrated in this section, the scientific and especially grey literature[6] provide a range of examples of how international crises, specifically armed conflicts and emergency relief, may act as a driver facilitating biological invasions both through intentional introductions for various purposes and unintentional introductions through contaminants and stowaways. Authors found evidence from plants, vertebrates, invertebrates and microorganisms; from terrestrial, aquatic and marine systems; and from all IPBES regions. The literature on international crises and aid as a driver facilitating biological invasions covers all stages of the biological invasion process, especially focusing on the introduction and spread phases, the latter often from assessments of impact (**Chapter 4**).

3.2.2.4 Urbanization

Urbanization, the increase in the proportion of a population living in urban areas, results in a large number of people becoming permanently concentrated in relatively small areas, forming cities (IPBES, 2022c). By 2018 approximately 55 per cent of the world's population resided in urban areas, and it is expected that this will exceed 60 per cent by 2030 (United Nations, Department of Economic and Social Affairs, Population Division, 2019). As the global human population increases, the patchwork of urban sprawl and modified environments is increasingly dominating landscapes. This accelerated urban growth has contributed to the extensive fragmentation, reduction and degradation of natural ecosystems worldwide (IPBES, 2019) facilitating the establishment and spread of invasive alien species in urbanized areas (**sections 3.3.1.2** to **3.3.1.5**). Along with the increase in human population density in cities (**section 3.2.2.1**), the volume, frequency and range of movement of people and goods also increases, as does trade (**sections 3.2.3.1** to **3.2.3.4**). The movement in people and goods operates at many scales, from local, through national and regional scales, to global networks, facilitating transport of invasive alien species at all scales.

In a global review of invasive alien plants, vertebrates and invertebrates on islands (749 alien species in total), urban areas had consistently higher abundances of alien species compared to natural ecosystems (Sánchez-Ortiz *et al.*, 2020). Urban areas are centres of transport and travel, facilitating breakdown of biogeographic barriers and high rates of introduction of invasive alien species into urban habitats (Banks *et al.*, 2015). Accordingly, the richness of alien plants in urbanized areas is related to factors such as length of railroads and roads, and the size of the urbanized area (Kühn *et al.*, 2017). However, pathways vary across taxonomic groups, with escape from containment in homes or gardens being the most likely source of invasive alien plants and vertebrates in urban areas, whereas invertebrates are likely to arrive as stowaways or contaminants in transported goods (Padayachee *et al.*, 2017). Many invasive alien species have generalist or opportunistic traits, and urban areas may provide suitable environments and novel opportunities for their establishment and spread. In the case of birds, for example, urban environments offer opportunities for species with flexible foraging strategies to adopt novel food sources; favouring invasive alien birds that tend to be more flexible in their behavioural traits relative to native species (Griffin *et al.*, 2017). Similarly, for plants, the increased disturbance (**section 3.3.1**), pollution and nutrient availability (**section 3.3.3**), and climate warming (**section 3.3.4**) associated with urbanization generate opportunities for alien species, many of which are habitat generalists and/or disturbance and high-nutrient habitat specialists. The number of invasive alien species in urban areas is predicted to increase further due to an increase in propagule pressure and opportunities for the spread and establishment associated with increased global trade, intentional release of alien species, land-use intensification, urbanization and climate change (Dawson *et al.*, 2017).

In some cases, urban areas are a focus for the spread into the wider environment and may have major implications for Indigenous Peoples and local communities, as in the case of Montréal, where wetlands are invaded by invasive alien species due to clearing and filling activities occurring around Indigenous lands (IPBES, 2022b). Accordingly, many studies have found a positive association between the number of invasive alien species within a location and the percentage of urban land in the surrounding landscape (Vilà & Ibáñez, 2011). Similarly, road density, frequency of road use and road improvement increases diversity of alien species in adjacent ecosystems (Vilà & Ibáñez, 2011).

Urbanization can also be a driver for aquatic invasive alien species. Out of 891 species listed in the Global Invasive Species Database (GISD; GISD, 2021), 277 (31 per cent) are associated with urban areas, 395 (44 per cent) are associated with inland waters (wetlands, lakes, or water courses), and 147 (16 per cent) are associated with both urban areas and inland waters (**Chapter 2**, **section 2.5.5.1**). Urbanization provides two means for enhanced invasions into wetlands. First, large numbers of species are imported (intentionally or unintentionally) into urban areas, creating high propagule pressures, and second, urbanization causes disturbance of existing ecosystems (Hassall, 2014). For example, deployment of infrastructure (**section 3.3.1.4**) such as stormwater ponds may harbour the invasive

6. Data management report available at https://doi.org/10.5281/zenodo.5529309

alien *Anaxyrus fowleri* (Fowler's toad) and construction may facilitate the dispersal of this invasive alien species. Urbanization may facilitate biological invasion in or around aquatic environments through modification of channels and banks (**section 3.3.1.4**), disturbance from traffic, presence of pet animals (**section 3.3.5**), dumping of rubbish (**section 3.3.3**), and reductions in permeability of surrounding land (Hassall, 2014; **section 3.3.1.6**).

Urbanization also plays a role in the relative distribution and abundance of both native and alien species in marine environments. Deployment of marine infrastructure (**section 3.3.1.4**) is an important direct driver, and large coastal and marine areas of Europe, North America, Asia and Australia are nowadays covered by sea walls, dikes, breakwaters, groynes, jetties, pilings, bridges, artificial reefs, offshore platforms and energy installations, which are linked to urban areas. A study carried out by Airoldi *et al.* (2015) found that marine infrastructures along sandy shores disproportionally favour alien over native hard bottom species, affecting alien species' spread at regional scales.

3.2.3 Economic drivers

Alien species are dispersed globally and regionally both intentionally and unintentionally through trade and commerce (**sections 3.2.3.1** to **3.2.3.3**) and by travellers and tourists (**section 3.2.3.4**). While the causal relationships are complex, the resulting pattern can be relatively simple and predictable. The number of alien species found in a country often correlates with per capita GDP (Dawson *et al.*, 2017; Essl *et al.*, 2011) reflecting the intensity of international trade which is a major conduit for the introduction of alien species (Hulme, 2009).

Economic drivers impact and interact with other indirect drivers of change in nature, especially demographic drivers, such as population density (**section 3.2.2.1**) and urbanization (**section 3.2.2.4**). Economic drivers operate by influencing the rate and magnitude of change in the majority of the direct drivers of change in nature, including those related to land- and sea-use change (**section 3.3.1**), direct exploitation of natural resources (**section 3.3.2**), pollution (**section 3.3.3**), climate change (**section 3.3.4**), and drivers related to the intentional introduction of invasive alien species and associated unintended consequences (**section 3.3.5.2**) and biodiversity loss (**section 3.4.2**). Section 3.2.3 describes evidence for links between specific economic drivers and invasive alien species (**Figure 3.9**) and makes reference to other indirect and direct drivers when relevant. Mechanistic links between the ensuing direct drivers and invasive alien species are discussed in **section 3.3**.

3.2.3.1 International trade and global commerce

International trade (whether legal or illegal) in commodities (e.g., minerals, petrochemical products, agricultural products, machinery and electronic goods, plants and wildlife) is an important route through which invasive alien species are introduced into new regions (Hulme, 2021b). International trade has grown dramatically since 1950 (Hulme, 2009, 2021b), and the quarterly world trade volume more than doubled from 2005 (2.5 trillion US dollars (US$)) to 2019 (over US$6 trillion; WTO, 2021), so that few nations in the world are not linked to others through trade. In many ways, trade is a universal driver facilitating biological invasions across contexts, playing a role in the introduction of aquatic and terrestrial taxa

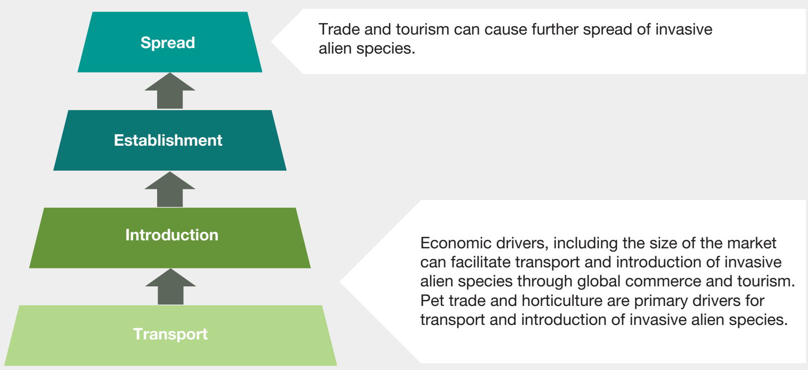


Figure 3 9 **Examples of the role of economic drivers in facilitating invasive alien species across stages of the biological invasion process.**

Economic factors such as size of markets and trade and use of specific invasive alien organisms can facilitate their transport and introduction, whereas trade and tourism can facilitate the further spread of invasive alien species within regions.

across the world. The demand for increased international trade has led to a major shift in the magnitude and reach of international shipping, and to the development of ports and deep-water harbours, inter-regional canals and global air freight (Hulme, 2009a; **section 3.3.1.3**, **Box 3.7**; **section 3.3.1.4**). Government officials at international borders regularly intercept invasive alien species associated with a wide range of imported commodities and transport vectors (Bacon *et al.*, 2012; Caley *et al.*, 2015; McCullough *et al.*, 2006; Work *et al.*, 2005). Clearly, the more locations at which international commodities first arrive in a country (airports, seaports, land-borders), the greater the likelihood that invasive alien species will succeed in finding a suitable environment (**Chapter 1**, **section 1.4.1**; **Chapter 2**, **section 2.1.2**).

The direct effects of trade have largely been quantified using relationships between imports and the number of alien species in a region or patterns in the global spread of species linked to shipping and air traffic networks (Hulme, 2021b). Alien species may themselves be the imported commodity (e.g., aquarium, ornamental, pet, crop, or pollinator species; **Box 3.3**), or a contaminant of a commodity (e.g., plant pathogens on a host plant, seeds trapped in wool fleeces, or insects in grain shipments), or be associated with a transport vector as a stowaway (e.g., hull fouling or ballast water organisms associated with marine transport). For example, invasive alien species of ornamental fish (Padilla & Williams, 2004; Hixon *et al.*, 2016), plants (Hulme *et al.*, 2018) and insects **(Box 3.3)** are traded globally as a commodity. The international pet trade is recognized as a primary driver for the introduction of invasive alien animals (Hulme, 2015a; Maceda-Veiga *et al.*, 2013). The introduction of *Aedes albopictus* (Asian tiger mosquito) is well known as a contaminant of imported used tyres (Benedict *et al.*, 2007), and large numbers of alien vertebrate and invertebrate species have been introduced as stowaways in ballast water (Molnar *et al.*, 2008). There is also evidence that illegal wildlife trade also runs the risk of introducing some invasive alien species into new regions (García-Díaz *et al.*, 2016). Furthermore, international trade has been a driver for the initial construction and, more recently, expansion of shipping canals (e.g., Suez, Panama) that by linking previously separate marine regions have facilitated the spread of invasive alien species between seas and oceans (Golani, 2021; Hulme, 2015b; **section 3.3.1.3**, **Box 3.7**). In addition to the transport and introduction of invasive alien species across international borders, commerce can also facilitate the establishment and spread of alien species within a region. For example, the size of the national market for ornamental plants (Dehnen-Schmutz *et al.*, 2007) and freshwater crayfish (Chucholl, 2013) is a strong driver that increases the likelihood of species becoming established in the wild. Such trends are likely to be exacerbated by the growth in e-commerce (**Glossary**) of alien species (Humair *et al.*, 2015).

Not all alien species introductions are associated with a specific commodity; some alien species may enter a region on vectors rather than commodities, that is they are associated with the mode of transport. Alien species introduced in ballast water are more likely to reflect the volume, frequency, age and origin of marine vessels than the specific commodities carried by shipping (Hulme, 2021b). The network of global ship movements and the estimated volume of ballast water discharges in ports worldwide have been used successfully to identify the major source regions for invasive alien species in several maritime ecoregions (Seebens *et al.*, 2016). However, there has been a five-fold increase over the past 30 years in the number of shipping containers carrying international trade but, despite the risk containers pose, there have been no studies to date attempting to relate contemporary risk of invasive alien species to variation in the number of containers imported or indeed their global itineraries (Hulme, 2021b).

There is often a strong correlation between the number of alien species in a country and the value of commodity imports, supporting the view that international trade is a key driver in the introduction of invasive alien species (Levine & D'Antonio, 2003; Santini *et al.*, 2013; Seebens *et al.*, 2015; Westphal *et al.*, 2008). Such relationships are often nonlinear, suggesting that the effect of imports on alien species numbers becomes less strong once a certain threshold is reached, and can vary in strength quite markedly depending on the taxonomic group examined (Hulme, 2021b). Coarse correlations with the value of all commodity imports often mask important detail regarding the relationship between trade and the multiple pathways of introduction of alien species (Hulme *et al.*, 2008). Even within a single pathway there may be subtle differences in the risk posed by particular commodities, as in the case of invasive alien insects that are more likely to enter the United States as contaminants of commodities transported in refrigerated rather than unrefrigerated cargo (Work *et al.*, 2005).

Knowledge that the likelihood of the introduction of alien species varies by trading partner has led to studies examining wider bilateral trade relationships more thoroughly (Hulme, 2021b). As a result, data on the magnitude of bilateral trade between regions is often used to estimate the scale of future biological invasions and has, to date, pointed to a significant increase in risk over the next decades (Bradley *et al.*, 2012; Seebens *et al.*, 2015). These risks differ by global region, with some evidence suggesting there are greater risks for developing countries (Seebens *et al.*, 2015). Given the strong link between international trade and biological invasions, the global economic slowdown that was initiated by the 2008 financial crisis (Constantinescu *et al.*, 2016) might be expected to have reduced the rate of alien species establishment however, evidence suggests rates of establishment are likely to lag behind economic drivers by as much as two decades (Seebens *et al.*, 2015). Thus, for the foreseeable future, rates of alien species introductions through trade will continue to increase. As a consequence, future trends in international trade, including e-commerce, new trade routes, and major infrastructure developments, will lead to pressure on national borders that may soon outstrip the resources available for intervention (**Chapter 6**, **section 6.3.1.4**).

Box 3 3 **Trade of bumblebee colonies for crop pollination as a driver that facilitates the introduction of invasive alien species.**

Many pollinators and flower visitors from various insect orders and families have been introduced and established out of their native ranges (Bartomeus *et al.*, 2013; Goulson, 2003, **Chapter 1**, **Box 1.11**). Unintentional transportation with their host materials accounted for the establishment of cavity-nesting bees (e.g., *Anthidium manicatum* (wool carder bee)), whereas crop pollination motivated the intentional transportation and introduction of *Apis* (honey bees) and *Bombus* (bumble bees) (Gibbs & Sheffield, 2009; Goulson, 2003; Schweiger *et al.*, 2010). While different subspecies of *Apis mellifera* (European honeybee), native to Europe and Africa, have been managed at least since 2450 B.C., (Crane, 1999) and introduced in all continents (except Antarctica) where they were not native, the introduction of *Bombus* spp. (bumblebee) colonies is relatively recent (Osterman *et al.*, 2021). Bumblebees live in colonial nests and can buzz pollinate (whereby bees use vibrations to extract pollen from flowers, incidentally, fertilizing them), making them suitable pollinators for a wide range of crops, in particular those grown in greenhouses. The rearing of bumblebee colonies of European species *Bombus terrestris* started in the 1980s and in a few years triggered a massive trade of colonies within and beyond its native range (**Figure 3.10**). This species has invaded many countries in which it has been intentionally introduced (e.g., Japan, New Zealand, Chile) expanding its range even to countries were introduction was not allowed, as is the case of Argentina (Aizen *et al.*, 2019). In South America, *Bombus terrestris* has spread across Chile (Montalva *et al.*, 2011) from the Atacama Desert to the southernmost islands south of the Tierra del Fuego Archipelago and to South West Argentina (Morales *et al.*, 2013). In this region *Bombus terrestris* achieves unusually high abundance and dominance of local pollinator communities (Aizen *et al.*, 2014; Morales *et al.*, 2017). Moreover, the indroduction of *Bombus terrestris* has facilitated the co-introduction of novel pathogens (Arbetman *et al.*, 2013; **section 3.3.5.1**).

Figure 3 10 **Growers can purchase bees in a box that will fly from flower to flower, distributing pollen among the plants.**

Many pollinators and flower visitors from various insect orders and families have been introduced and established out of their native ranges. Photo credit: jpr03, Adobe Stock – Copyright.

3.2.3.2 Trade in plants for horticulture, agriculture, ornamental use and nurseries

Globally, 67 per cent of alien terrestrial plants have been introduced intentionally through horticultural (46 per cent) or agricultural (21 per cent) pathways (Turbelin *et al.*, 2017). Ornamental use of plants is a dominant driver facilitating the introduction of invasive alien species that has also been increasing in recent years (Dodd *et al.*, 2015; Faulkner *et al.*, 2016; Hulme *et al.*, 2018; Lambdon *et al.*, 2008; Lehan *et al.*, 2013; Mayer *et al.*, 2017). Accordingly, there is an increasing rate of escapes from ornamental cultivation into the wild (Haeuser *et al.*, 2018; van Kleunen *et al.*, 2018), with at least 75 per cent of the global established alien flora grown in domestic gardens, and 95 per cent grown in botanical gardens (van Kleunen *et al.*, 2018). In some countries (e.g., United Kingdom, New Zealand) the number of species in cultivation exceed the number of native species in the wild (Armitage *et al.*, 2016; Gaddum, 1999; Hulme, 2020), and in some, such as the United States, alien species form the bulk of nursery stock (Brzuszek & Harkess, 2009). Unintentional introduction through seed contaminants associated with the intentional introduction of ornamental plants is regarded as the second most important source of invasive alien plants in the United States and has become increasingly important in recent years (Lehan *et al.*, 2013). Horticulture was also the primary pathway (**Glossary**) of alien introductions in Puerto Rico and the Virgin Islands (Rojas-Sandoval & Acevedo-Rodríguez, 2015). In the West Indies, 75 per cent of all invasive alien plants have escaped from cultivation, and 51 per cent of all introductions are through the ornamental pathway (Rojas-Sandoval *et al.*, 2017).

Ornamental and agricultural use is also a major source of aquatic plant invasions. In the People's Republic of China, more than 50 alien freshwater aquatic plants have been introduced for ornamental, landscaping, water purification, forage and other purposes, around 20 per cent of which are now considered invasive (Wu & Ding, 2019). In Europe and the Mediterranean region, large numbers of ornamental aquatic alien plants are unintentionally released from aquaria, dumped from water gardens, or escaping from managed environments, a number of which have become invasive (Brundu, 2015). Many ornamental aquatic alien species may become widely distributed, for example *Pontederia crassipes* (water hyacinth) which was introduced from South America to botanic gardens and ornamental ponds around the world from the nineteenth century onwards is now found across more than 50 countries on five continents (Sharma *et al.*, 2015; **section 3.5.2**, **Box 3.12**).

3.2.3.3 Trade in terrestrial pet animals

Pet trade is a major pathway of alien animal introductions (Hulme, 2015a). It is estimated that 70 per cent of households in the United States and 38 per cent of households in Europe have pets. In addition to the majority of dogs and cats, fishes, birds, small mammals and reptiles are commonly kept (Mazzamuto *et al.*, 2021). In Europe, a systematic review revealed invasive alien mammal introductions were primarily a result of escapes by pets (69 per cent) and from zoos (50 per cent) and fur farms (38 per cent), while far fewer arose from other agricultural species or biological control and none were reported as contaminants, stowaways or *via* corridors (Tedeschi *et al.*, 2021). Pet escapes and releases are major drivers facilitating vertebrate invasions (e.g., mammals in Brazil (da Rosa *et al.*, 2017) and amphibians and reptiles of the United States (Krysko *et al.*, 2016) and European Union (Hulme *et al.*, 2008; Katsanevakis *et al.*, 2013)).

Pets also have the potential to become important vectors for pathogens and microorganisms that cause disease in animals and humans, in particular pets derived from wild animals (Day, 2011). The spread of monkeypox to humans in the United States is thought to be due to contact with prairie dogs sold as pets (Brown, 2008) and pets are considered to have been important in the transmission of chytridiomycosis and ranaviral disease, which cause severe damage to amphibians (Schloegel *et al.*, 2012).

3.2.3.4 International travel for commerce and tourism

In 2019, there were approximately 1.5 billion international passenger arrivals associated with tourism, a five-fold increase in the number of travellers compared with 1980 (UNWTO, 2021). The continued expansion of the worldwide air transport network has facilitated this global movement of human passengers and, by increasingly linking regions of the world with similar climates, has facilitated the introduction of invasive alien species (Tatem, 2009). While some invasive alien species, particularly mosquitoes (Brown *et al.*, 2012), can be unintentionally transported in commercial passenger aircraft, the highest risk of introducing a wide range of alien species comes from the passengers. Passengers often intentionally transport fresh food items, untreated timber, or animal skin products, either for personal consumption or as gifts, which may carry alien species or alien pathogens as contaminants on the passengers themselves or in their luggage. In the United States, border inspections have found that more than half of all pests encountered were associated with traveller baggage rather than cargo (McCullough *et al.*, 2006). Although some international travellers attempt to intentionally smuggle live animals, plants and food products that could be themselves invasive alien species or harbour them as contaminants (Chown *et al.*, 2012; Soon & Manning, 2018), most international travellers are unaware of the risk they pose in unintentionally introducing stowaways. International travellers can also introduce stowaways on their clothing, footwear and equipment (e.g., tents, fishing tackle and golf clubs). A single gram of soil attached to footwear can harbour 5 million bacteria, 50,000 fungi, 3 seeds and 40 nematodes, and these taxa may often include potential invasive alien species (Hulme, 2015a). When these figures are multiplied up by the more than one billion international travellers worldwide (Glaesser *et al.*, 2017), the global movement of alien stowaways is substantial. Up to half of hikers sampled in mixed evergreen forest in California were found to be carrying *Phytophthora*

Box 3 4 International tourists and scientists visiting Antarctica.

Strong climatic and geographic barriers, attributes associated with low habitat invasibility, naturally isolate Antarctica from the rest of the world (Chwedorzewska *et al.*, 2020). Climate change and an increasing number of visitors (G. A. Duffy *et al.*, 2017; Greve *et al.*, 2017) have weakened these barriers, leading Antarctica to become an area of special concern for the management of biological invasions (**Chapter 6**, **Box 6.10**). An average of 9.5 seeds per visitor are carried to Antarctica every summer (Chown *et al.*, 2012). The highest risk of seed transportation is associated with science programs **(Figure 3.11)** and tourist support personnel rather than with the increasing tourist numbers (Chown *et al.*, 2012; Chwedorzewska *et al.*, 2020; Huiskes *et al.*, 2014). Between the summers of 2019 and 2020, about 74,400 tourists visited Antarctica and 18 nations had established over 50 research stations (Hughes *et al.*, 2020; IAATO, 2020). Alien plants and invertebrate species in Antarctica are found almost exclusively close to visitor sites and research stations (Molina-Montenegro *et al.*, 2012, 2014).

Figure 3 11 **Research stations in Antarctica increase the risk of biological invasions.**

The Australian Antarctic Division has 4 permanent research stations in Antarctica and the subantarctic. Photo credit: David Barringhaus/Australian Antarctic Division – Copyright.

The introduction, establishment and spread of *Poa annua* (annual meadowgrass) in different localities in Antarctica is well documented (Chwedorzewska *et al.*, 2015). *Poa annua*, an annual grass native to Europe and an invasive alien species in the Andes, has been observed in the vicinity of Arctowski Station on King George Island in Antarctica since 1985 (Chwedorzewska, 2008). In this site, *Poa annua* maintains a genetically diverse population, which has been attributed to intense human activity in the station facilitating multiple introductions from different source populations (Chwedorzewska, 2008). Fuentes-Lillo *et al.* (2017) found a connection between human activity and invasive alien plants in the Fildes Peninsula in King George Island. These authors found higher concentration of seeds in soil samples at sites with increased human activity (i.e., next to dormitories of logistics personnel). Six of the eight alien species recorded in soil samples were also found in King George Island; *Taraxacum officinale* (dandelion) and *Poa annua* have been the most successful colonising alien species in these ecosystems (Fuentes-Lillo *et al.*, 2017).

ramorum, the causal agent of sudden oak death, in soil on their shoes (Davidson *et al.*, 2005). It also appears that tourists visiting caves may be responsible for spreading *Pseudogymnoascus destructans* (white-nose syndrome fungus) in bats (Puechmaille *et al.*, 2011). Similarly, tourists are known to spread weed seeds into national parks (Pickering & Mount, 2010).

The motives and Interests of tourists are also changing with increasing interest in recreational activities (e.g., golf, fishing), agritourism (e.g., winery visits), visits (including camping) to national parks and reserves (Hulme, 2015a). This change in behaviour poses an increased risk of introductions into areas that may not yet been exposed to invasive alien species. The number of passengers embarking on world and extended length cruises is doubling every decade (Klein, 2011) and has permitted access to coastal areas previously exposed to low numbers of visitors. Since 1990, international visitor numbers to the Antarctic continent have increased almost 10-fold (Hulme *et al.*, 2012) and up to half of tourists and visiting scientists unintentionally bring with them seeds of alien plant species which could pose a considerable risk of establishing in the region (Huiskes *et al.*, 2014). Much of this risk is not simply from the tourists themselves but the support crews **(Box 3.4)**. Similar risks are found in the Arctic where people and their luggage are responsible for around 5 per cent of all alien plant introductions (Wasowicz *et al.*, 2020). While the number of international passenger arrivals world-wide has more than doubled since 1990, it is in emerging economies in Africa, Asia and South America where the rate of growth of passenger arrivals has been highest and these regions may be less well prepared to face new risks from invasive alien species (Glaesser *et al.*, 2017). With forecasts of global tourist numbers reaching 1.8 billion international travellers in 2030, combined with new destinations previously less exposed to invasive alien species and more activities in less visited areas, the future risk of introducing invasive alien species appears significant (Hulme, 2015a).

3.2.3.5 Externalities of negative impacts and cost

The introduction of invasive alien species is usually an unintended or intended consequence of economic activities that not only brings species into areas where they were not present, but also affects the frequency of repeated introductions and the spread of established alien species (Touza *et al.*, 2007; **section 3.2.3**). However, control is difficult because those whose actions result in the introduction of invasive alien species are rarely those affected, and they are often not held accountable for their actions (Perrings *et al.*, 2005; Tollington *et al.*, 2017; **section 3.2.1**). For example, in trade of plants and animals, the prices paid by the importer to the exporter include production and transport costs, but usually do not include the costs incurred when animals or plants become invasive. Alien species that lead to improved quality of life and economic benefits are considered public goods, especially in cases where awareness of the impacts of invasive alien species is limited. In addition, the characteristics of public goods, such as “non-rival” and “non-excludable”, create incentives to free-ride and further increase market inefficiency (Marbuah *et al.*, 2014; Touza *et al.*, 2008). These activities of economic agents that affect others without going directly through the market are called “externalitie” (**Glossary**), and externalities that negatively affect other agents are called “negative externalitie” (**Chapter 6**, **section 6.2.1(6)**). The damages and management costs incurred by biological invasions are a typical case of negative externalities arising from economic actions, and for many of the examples listed under **sections 3.2.1**, **3.2.2.3** and **3.2.3.2**, negative externalities are an important explanation for why high rates of invasions are being sustained (**Chapter 2**) despite known negative impacts on nature, nature’s contributions to people and good quality of life (**Chapter 4**).

Internalization of externalities can be a powerful driver to prevent the introduction of invasive alien species (**Chapter 6**, **section 6.2.1.(6)**). But the current situation of leaving externalities unattended without recognizing their existence is a driving force for the introduction of invasive alien species that underlies many social, economic and demographic drivers.

3.2.3.6 Wealth, inequality and poverty

Are invasive alien species primarily a problem of wealthy countries? National GDP has been a frequent metric used to explain variation in the number of alien species among different countries, with countries having a higher GDP being more invaded (Essl *et al.*, 2011; Hulme, 2021b; Seebens *et al.*, 2017). This positive relationship between GDP and the number of alien species in a country appears to hold both across global and regional scales as well as for quite different taxa including fish, mammals, birds, plants and agricultural pathogens (**Chapter 2**, **section 2.1**). Such trends may be indicative that countries with higher levels of consumption tend to facilitate the introduction and establishment of alien species. However, the current number of alien species in a country is the result of a cumulative process where alien species have accumulated over several centuries (**Chapter 2**). Hence, research has shown that historical levels of GDP might be a better predictor of the number of alien species found in a country than the current GDP (Essl *et al.*, 2011). This illustrates that GDP is a flow and that the best measure of a process that, in many countries, has deep historical roots, is the stock of wealth in any country. This wealth measure is the cumulative effect of past investments, and accounts for the assets such as natural capital, produced capital, and human capital that underpin growth and consumption possibilities (Perrings, 2010). Few studies have examined the role of wealth on numbers of invasive alien species, but where it has been examined it has been found to explain a small but significant amount of variation in the numbers of alien species in Europe (Pyšek *et al.*, 2010). Since the wealth of a country is a more important determinant of numbers of alien species than the contemporary GDP, the cumulative build-up of assets which support greater consumption may lead to more immediate increases in numbers of alien species than rapid changes in GDP.

The current combined gross national income of all countries in the world is estimated to be over US$94 trillion, with 10 countries accounting for 68 per cent of this amount (IMF, 2021). The distribution of this wealth is also often highly uneven within countries, so that the 10 per cent richest individuals concentrate 52 per cent of the world income, whereas 50 per cent of the global population accounts for only 8.5 per cent (Chancel *et al.*, 2022). Poverty and marginalization created by economic inequality within and among countries may indirectly drive the introduction, establishment and spread of invasive alien species. For those countries with a lower level of wealth, the warning signs suggest that as the economies grow and build a larger asset base, the risk of alien species introductions might also increase. This risk may be further exacerbated where the route to economic growth and poverty reduction encourages the development of economic sectors based around alien species (**sections 3.2.2.3**, **3.2.3.6**, **3.2.5**, **3.3.1.1**). For example, the invasive alien tree *Prosopis juliflora* (mesquite) was intentionally introduced in many eastern Africa countries to improve the livelihood of communities with very low income subjected to malnutrition and food shortages, as it may be a reliable source of firewood and animal fodder in arid regions (Pasiecznik *et al.*, 2007; **section 3.2.5**, **Box 3.6**); however, the species have become highly dominant in many regions, thus decreasing biodiversity and threatening the water supply (Pasiecznik *et al.*, 2007; **Chapter 4**, **boxes 4.8** and **4.9**). Invasive alien species are also important for the subsistence and income of certain low income communities in South Africa, including the invasive cactus *Opuntia ficus-indica* (prickly pear) in the Eastern Cape Province and multiple alien species cultivated in urban gardens in the Limpopo province (e.g., the invasive tree *Schinus terebinthifolius* (Brazilian pepper tree)). In many cases, it is not clear how the cultivation of these invasive alien species has contributed to their spread (Mdweshu & Maroyi, 2020; Mosina *et al.*, 2015). Some Indigenous Peoples and local communities can also be more inclined to use alien plants if they experience a loss of traditional medicine knowledge and a loss of historically used medicinal plants (IPBES, 2020). In rural Mexico, there are economic incentives to farm the invasive alien fish *Oreochromis* spp. (tilapia) which has been viewed as having a positive impact as a route out of poverty (Martinez-Cordero & Sanchez-Zazueta, 2022). However, there are examples showing that using alien species to promote economic growth may lead to more poverty and inequality, such as the use of *Lates niloticus* (Nile perch), which was introduced into many African lakes (Kelly, 2018; see **section 3.2.5, 3.3.2.1**). In the Lake Victoria basin, the invasion of the Nile perch nearly exterminated native fish populations, and local communities were ultimately forced to shift from traditional fishing, which ensured their subsistence, to catching the invasive alien perch. Nile perch fishing supplies external markets and provides minimum income to households, thus leading to more poverty, malnutrition and lower quality of life (Geheb *et al.*, 2008; **Chapter 4**, **Boxes 4.8** and **4.9**).

3.2.4 Science and technology

While science and technology are major factors underpinning demographic and economic changes, science and technology can also act as indirect drivers of change in nature affecting invasive alien species **(Figure 3.12)**. This section focuses on how research activities (**section 3.2.4.1**), the rise and spread of communication technology (**section 3.2.4.2**), and breeding and genome technologies (**section 3.2.4.3**) can facilitate the transport, introduction, establishment and spread of invasive alien species. Science and technology may interact with economic drivers through the role of information technology in supporting international trade in invasive alien species (**section 3.2.3.1**), and with biological invasions *via* unintended consequences of introducing and/or controlling invasive alien species themselves (**section 3.3.5**). Section 3.2.4 describes evidence for links between specific science and technology drivers and invasive alien species, and makes reference to other indirect and direct drivers when relevant, whereas the specific contribution of science and technology to controlling invasive alien species is dealt with in **Chapter 5**, **section 5.4.4**.

3.2.4.1 Research

Scientific research can involve the transport, rearing, storing, manipulation and experimental release (either under controlled conditions or outdoors) of living organisms. While most evidence to date points to scientific research and related activities as important drivers in the stages of transportation and introduction of invasive alien species, there is a paucity of evidence on their role in establishment and spread.

There are a number of documented examples of use of organisms outside of their native ranges for laboratory research which have resulted in biological invasions. For instance, *Xenopus laevis* (African clawed frog) is the most studied amphibian worldwide and one of the best model organisms for studies in cell, molecular and developmental

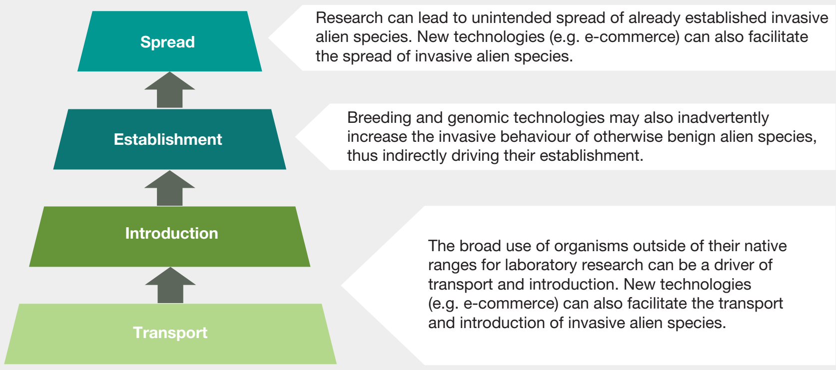


Figure 3 12 **Examples of the role of science and technology in facilitating invasive alien species across stages of the biological invasion process.**

Science and technology can facilitate the transport and spread of invasive alien species *via* their specific use in research and their trade through e-commerce. Breeding and genome technologies may select for traits that increase invasiveness, and research and e-commerce can also facilitate further spread of invasive alien species.

biology. *Xenopus laevis* has been continuously introduced for the past 50 years, and intentionally released in Europe, Asia and North and South America and is likely responsible for the spread of the chytrid fungus of the genus *Batrachochytrium* (Fisher & Garner, 2007, 2020; Weldon *et al.*, 2004). Agronomic research has also led to the introduction of wild relatives of crops outside their ranges. For instance, populations of *Solanum chacoense* (Chaco potato), a wild relative of *Solanum tuberosum* (potato), has established close to research centres where the species was most certainly introduced as part of breeding programs in Australia, China, New Zealand, United States and Argentina (Simon *et al.*, 2010). Unintentional or intentional releases from aquaria have been listed among the major sources of invasive alien invertebrates and fishes in estuaries and rivers (Englund, 2002), and escapes of plant propagules or seeds from botanic gardens is also responsible for some major plant invasions **(Box 3.5)**. In addition, unintentional releases from experimental farms conducting agricultural research (**section 3.3.1.1.4**) are major sources of some of the established populations of highly invasive mammals and birds including *Mustela vison* (American mink), *Myocastor coypus* (coypu), *Ondatra zibethicus* (muskrat), *Nyctereutes procyonoides* (raccoon dog), *Procyon lotor* (raccoon), *Threskiornis aethiopicus* (sacred ibis) and *Oxyura jamaicensis* (ruddy duck) (Barrat *et al.*, 2010).

Devices, sampling gear and equipment used in research activities within natural habitats may act as vectors of invasive alien species. For instance, barnacles frequently attach to the leg-rings used to identify individual wading birds by ornithologists and can be transported long distances along bird migratory routes, "hitch-hiking" with their host birds. Since more than 30 living barnacles can attach to a single ring, this can lead to substantial transportation and introductions to new areas (Tøttrup *et al.*, 2010). More recently, the use of submersible assets, like remote-operated vehicles and human-occupied vehicles, open a novel potential pathway through which scientific research can aid the transport of invasive alien species. In particular, the use of remote-operated vehicles has expanded the reach of human influence to regions where humans themselves cannot access. These high-technology vectors have a tremendous potential to increase transport of invasive alien species in marine ecosystems (Thaler *et al.*, 2015), as illustrated by the transport and potential introduction of living limpets from deep-sea hydrothermal vents to other distant vents (Voight *et al.*, 2012).

3.2.4.2 Development of communication technology

The development of communication technology is an important driving force for the globalization of markets. As of 2021, 63 per cent of the world population (i.e., 4.9 billion people) had internet access, an almost five-times increase compared with 2005, and 95 per cent was covered by broadband mobile network (i.e., 3G and over; International Telecommunication Union, 2021). New distribution channels such as internet trading (e-commerce) have caused significant changes in the movement of organisms. In e-commerce trading surveys, between 30 and 80 per cent of recognized invasive alien plant species were detected on auction sites daily, making it possible to obtain invasive alien plants from almost anywhere in the world (Humair *et al.*, 2015). Due to the difficulty in tracing the contents of express mail, and the increasing volume of trade through this route, the current biosecurity (**Glossary**) system cannot

Box 3 5 **The role of botanic gardens in the introduction of invasive alien plants.**

Botanic gardens **(Figure 3.13)** have made a significant contribution to the collection, cultivation and distribution of plant species for research and scientific uses worldwide (Sharrock, 2011) but have also been implicated in the introduction, early cultivation and/or local spread of invasive alien plant species into global biodiversity hotspots (Hulme, 2011a). Studies exploring the global increase of alien plant species have identified the establishment of botanic gardens as an important driver facilitating biological invasions (Seebens *et al.*, 2017). The global emergence of new alien plant species, defined as the first record of an alien species anywhere in the world, has been found to be related to the number of botanic gardens established in a region (Seebens *et al.*, 2018). A study from China examined the association between the first record of an invasive alien species in a region and multiple possible explanatory factors, including botanic gardens (Ni & Hulme, 2021). The researchers found that botanic gardens, including their history and number of species in living collections, generally play more important roles in influencing the number of first records of invasive alien plants compared to socioeconomic variables (such as Gross Domestic Product) and environmental factors (such as climate). However, invasive alien species primarily introduced for horticultural uses were more influenced by botanic gardens, while those introduced for agricultural uses were more strongly associated with climate variables, and the numbers of species introduced unintentionally were shaped by trade (**sections 3.2.1**, **3.2.3.1** to **3.2.3.4**, **3.3.1.4**). This research is the strongest evidence to date pointing to the role botanic gardens may play in the invasions of ornamental plant species. Despite signing up to the Global Strategy for Plant Conservation, most botanic gardens rarely implement regional codes of conduct to prevent plant invasions, few have a policy for biological invasions, and there is limited monitoring (**Glossary**) of garden escapes (Hulme, 2015b). Given the rapid increase in living collections, especially in Asia, this suggests botanic gardens may become increasingly important for the introduction of alien species in the future.

Figure 3 13 **Botanical gardens facilitate the introduction of invasive alien species.**

The China National Botanical Garden cultivate 6,000 species of plant, including 2,000 kinds of trees and bushes, 1,620 varieties of tropical and subtropical plants, 500 species of flowers and 1,900 kinds of fruit trees several of which are believed to have escaped and become established. Photo credit: Top Photo Corporation, Shutterstock – Copyright.

adequately manage internet trade (Humair *et al.*, 2015). In e-commerce, invasive alien plants are traded over longer distances than traditional commerce in plants, where people visited horticultural nurseries to make their purchases. In particular, seeds of invasive alien plants were transported further than saplings of alien plants (Lenda *et al.*, 2014). In the aquatic alien plant trade, a variety of invasive or import prohibited plants were sold online, and the traders lack knowledge about species identification and regulation. This is also the case for the trade in freshwater aquatic plants for aquaria in Brazil, where unregulated e-commerce has stimulated illegal trade and contributed to the introduction and spread of invasive alien macrophytes (Peres *et al.*, 2018). In the internet trading of ornamental marine life, unregulated e-commerce has contributed to the introduction and spread of invasive alien macroalgae in Europe, North America and Australia (Walters *et al.*, 2006). Many wholesalers and retailers lack awareness of the potential risks of ornamental marine species (Morrisey *et al.*, 2011). Online trading of freshwater invasive alien crayfish as pets is prevalent and has become a major route of biological invasions of crayfish and their contaminants and stowaways in many parts of the world (Chucholl, 2013; Faulkes, 2010, 2015; Papavlasopoulou *et al.*, 2014). E-commerce and social media also have supported an extensive world trade network of alien ornamental freshwater fishes, where customers often also have little knowledge of the risks of releasing these invasive alien species in the wild (Magalhães & Jacobi, 2010; Mazza *et al.*, 2015).

3.2.4.3 Breeding and genomic technologies

The modification of phenotypic traits to increase biomass, growth, and resistance to pests, diseases and stressors, etc., can increase the invasion potential of species that otherwise would not be invasive (e.g., Flory *et al.*, 2012 and references therein). Plant and animal traits that have traditionally been modified through breeding, selection and hybridization can now be more effectively modified *via* genomic technologies. The application of these techniques to crop and stock species has catalysed debates over the invasive potential of these “novel taxa” and how to adapt invasive alien species risks assessments and biosafety measures to deal with them (Hoenicka & Fladung, 2006; Luke Flory *et al.*, 2012; Quinn *et al.*, 2015; **Chapter 5**, **section 5.4.4.2**).

Studies suggest that risks posed by novel taxa are often inevitable and can vary spatially, temporally and according to the type of organism and the purpose of the trait selection process (Ellstrand *et al.*, 2013). There is good evidence that traditional breeding of plant species native to other regions for ornamental purposes has increased their invasive potential in North America (Ross *et al.*, 2008; Wilson & Mecca, 2003). For example, cultivars of the ornamental Asian shrub *Ardisia crenata* (coral berry) artificially selected for dense foliage show higher competitive ability and seedling recruitment success than native populations, which likely favours their dominance in the understory of mesic forests in Florida (Kitajima *et al.*, 2006). The ornamental clonal herb *Kalanchoe × houghtonii* (Houghton’s hybrid), which is an artificial hybrid obtained from the crossing of two species endemic to Madagascar (*Kalanchoe daigremontiana* (devil’s backbone) and *Kalanchoe delagoensis* (chandelier plant)) that exhibits a highly effective clonal growth which possibly has contributed to its wide distribution and local dominance in tropical arid and semi-arid ecosystems (Guerra-García *et al.*, 2015; Herrando-Moraira *et al.*, 2020). There is concern that the invasive potential of some pasture grasses has increased as a result of intensive artificial selection, which has used multiple tools (e.g., ploidy manipulation and introduction of endophytes) to maximize grass productivity under the specific conditions of each region of interest (Driscoll *et al.*, 2014). Transgenes for herbicide resistance present in the genetically modified *Agrostis stolonifera* (creeping bentgrass) used as lawn grass were found in wild *Agrostis* populations, suggesting that the resulting novel genotypes may be persistent and possibly favour these genetically modified organisms outside cultivation (Reichman *et al.*, 2006). Similar or even higher risks may apply to the selection of bioenergy crops (Richardson & Blanchard, 2011); for example, selection for fertility in the highly productive *Miscanthus × longiberbis* (giant miscanthus cultivar) led to a high increase in its potential to escape cultivation (Smith *et al.*, 2015). Among invasive alien trees, selection for high timber and forage production may have indirectly increased the invasion potential of a cultivar of the tropical tree *Leucaena leucocephala* subsp. *glabrata* (white leadtree) in certain areas in Northeast Australia (C. S. Walton, 2003). Similarly, provenance forestry trials and common garden experiments designed to select the seed sources of alien pine species most likely to succeed under distinct conditions in the introduced range possibly contributed to the high invasion success of some of these species in South America (Zenni, 2014; Zenni *et al.*, 2017).

There is evidence that human selection of specific behaviours in the hybrid fish *Xiphophorus hellerii × maculatus* (red swordtail) contributed to the emergence of established invasive alien populations in Hawaii, with a behaviour syndrome characterized by high aggression and exploration (D’Amore *et al.*, 2019). Among invertebrates, the artificial hybrid between European and East African honeybee species shows superior pollen extraction ability and higher swarming rate (i.e., the process of colony splitting to generate new colonies) than the European species (Pesante *et al.*, 1987), which may have contributed to its spread in the Americas to the detriment of native pollinators (Santos *et al.*, 2012).

Cultivating genetically modified crops may also indirectly favour the spread of invasive alien pests and pathogens; for example, the invasive tomato moth, *Tuta absoluta* (tomato leafminer), has been found to establish and spread rapidly in farms planted with genetically modified tomato cultivars, whereas traditionally bred cultivars have been shown to be resistant to this pest (Rakha *et al.*, 2017). As an example of how artificial selection may indirectly favour vertebrate invasions, intensive artificial selection has increased the fecundity of domestic pigs, so that hybridization and admixture between populations of domesticated pigs and wild boar have been associated with feral swine genotypes with higher fitness and hence more likely to establish and spread in natural ecosystems beyond the species' range in Europe (Canu *et al.*, 2018; Fulgione *et al.*, 2016; Goedbloed *et al.*, 2013), North America (Smyser *et al.*, 2020) and South America (de Oliveira *et al.*, 2018).

3.2.5 Policies, governance and institutions

Policies, governance and institutions underlie most direct and other indirect drivers of change in nature in complex ways. For example, economic drivers (**section 3.2.3**) are strongly linked to policy and institutional drivers, which govern production through regulations, taxes and subsidies, and affect terrestrial, aquatic and marine bioproduction systems, which in turn facilitate biological invasions (**section 3.3.1**). A common thread running through many of these systems is that policies, governance and institutions are focussed on the economic and production systems, with consequences for biological invasions receiving little attention. These unintended consequences are the focus of this section, which deals with regulations, taxes and subsidies that may result in unintended facilitation of the introduction, establishment and spread of invasive alien species (**Figure 3.14)**. The role of policies and institutions explicitly tasked with the control of invasive alien species and management of biological invasions are addressed in **Chapter 6**.

In an era of globalization and increasing interconnectedness between people, states and regions, there is an increasing reliance on supranational arrangements for the organization of human societies, referred to as international institutions and organizations. The annual amount of international official assistance for development has increased from less than US$80 billion in 2000 to over US$127 billion in 2010, particularly as a result of increasing bilateral disbursements to low-income countries (World Bank & International Monetary Fund, 2012). By providing the regulatory frameworks for transboundary activities, international institutions and organizations may act as indirect drivers facilitating the uptake, transportation, establishment and spread of invasive alien species, due to promotion of other indirect demographic and economic drivers of change in nature such as migration (**section 3.2.2.2**), trade (**section 3.2.3.1** to **3.2.3.3**) and human travel (**section 3.2.3.4**). In addition, international institutions and organizations may also influence direct drivers of change in nature, including the deployment of infrastructure (**section 3.3.1.4**), introductions from agriculture, forestry, fisheries and aquaculture (**sections 3.2.3.2**; **3.2.3.3** and **3.3.1.1**), and multilateral measures against climate change which may again facilitate invasive alien species. Moreover, international agreements and supranational decisions may scale down to national governance, an indirect driver of change which may facilitate invasive alien species (**section 3.2.5**). Although the role of governance in facilitating invasive alien species has been occasionally evaluated (Evans *et al.*, 2018), with a few exceptions (Mwangi & Swallow, 2008; Pérez *et al.*, 2003), there is a paucity of studies specifically addressing the role of international organizations and institutions in managing biological invasions. While it is hard to draw overall geographical patterns because of the global scale of this driver, its influence is expected to be global. This section illustrates the issue of international institutions and organizations acting as an indirect driver promoting invasive alien species by facilitating different stages of the biological invasion process.

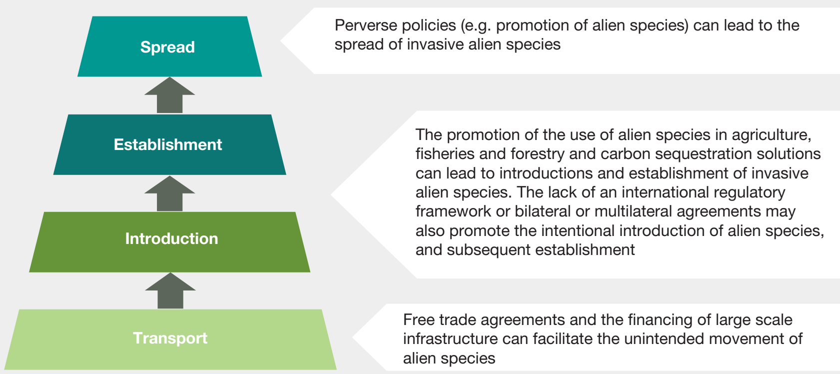


Figure 3 14 **Examples of the role of policies, governance and institutions in facilitating invasive alien species across stages of the invasion process.**

Free trade agreements, finance, promotion of species for use in agriculture, fisheries or forestry outside their native range, lack of regulatory frameworks and unintended consequences (active promotion of invasive alien species for other benefits, without consideration of risks) are some examples of drivers of changes in nature facilitating invasive alien species across invasion stages.

International funding agencies for economic development and regional integration may indirectly influence the uptake, and transport of invasive alien species by financing large scale infrastructure that facilitates the unintended movement of alien species (e.g., navigation channels and tunnels; Hulme, 2015b). In addition, international funding agencies may facilitate the introduction, release, and establishment of invasive alien species by actively promoting the use of alien species in agriculture, fisheries and forestry and carbon sequestration solutions (**sections 3.2.3.2**; **3.3.1.1**). An emblematic example is the promotion of the highly invasive leguminous tree *Prosopis juliflora* (mesquite; **section 3.2.3.6**; **Box 3.6**) from Central and South America into many arid and semi-arid regions of the world. Similarly, *Pontederia crassipes* (water hyacinth) has been promoted by international aid agencies in Africa for biomass production (**section 3.5.2**, **Box 3.12**). Its spread rapidly affected water flow regimes, impeding hydroelectric power production, water quality and fisheries (Batanouny & El-Fiky, 1984). In Vanuatu, the Central American woody species *Cordia alliodora* (Ecuador laurel) introduced to promote timber production, invaded native ecosystems and ultimately turned out to be unsuitable for timber production in that climate (Tolfts, 1997). Alien plants were introduced in many countries for economic development, including Tonga (Space & Flynn, 2001). *Lates niloticus* (Nile perch; **Chapter 4**, **Box 4.10**) is a notable example of an invasive alien species arising from aid-related fish introductions. This species was first introduced into Lake Victoria, East Africa, in the mid-1950s to supplement dwindling fish stocks. The population took 20 years to build up, but the Nile perch has since had a substantial impact on the ecological balance of the lake (Ogutu-Ohwayo, 1998), and is implicated in the extinction of more than 200 endemic fish species (Lowe *et al.*, 2000). Lake Victoria has been described as a major evolutionary and ecological disaster caused by the release of an invasive alien species, although the relative contributions and cause-effect relations between invasive alien species and other concurrent drivers of change in nature have been hotly debated (Marshall, 2018; van Zwieten *et al.*, 2015; **Chapter 4**, **Box 4.10**). More generally, international aid and assistance programmes have actively promoted the use of farmed plants, fish and animals in new regions, many of which are invasive and/or are hosts of other invasive alien species (**sections 3.2.2.3**; **3.3.1.1**).

Free trade agreements, being treaties between two or more countries or states to facilitate trade and eliminate trade barriers, increase the flux of goods between regions and may facilitate the uptake and transportation of invasive alien species, either as subject of commerce (when traded goods are living organisms), or more frequently as by-product of transport (e.g., unintentional introductions in ballast water, as stowaways in packaging and cargo, or as contamination of crop products with agricultural pests). Furthermore, the World Trade Organisation (WTO) in an attempt to prevent quarantine laws becoming trade barriers has discouraged nations from using quarantine laws to stop the spread of invasive alien species (Riley, 2005), illustrating the challenge of addressing national policies that drive biological invasions to account for the transboundary aspect of biological invasions (Hulme, 2015a).

More recently, as economic multilateral instruments have been developed to tackle climate change, taxes based on carbon and carbon-trading markets (e.g., Reducing Emissions from Deforestation and forest Degradation (REDD+)) have been implemented under the international governance framework of the Kyoto Protocol (United Nations, 1997). Such initiatives are a possible driver for the establishment and spread of invasive alien species, such as fast-growing trees from large-scale plantations for carbon

Box 3 6 National and international policies resulting in the introduction and spread of *Prosopis juliflora* (mesquite), as reported by Indigenous Peoples and local communities.

In Botswana, Ethiopia, India, Jordan and Kenya, Indigenous Peoples and local communities have reported the intentional introduction of *Prosopis juliflora* (mesquite) by governments and associated international programs with the aim of halting land degradation, controlling desertification and deforestation and improving the good quality of life of the local communities (Al-Assaf *et al.*, 2020; Becker *et al.*, 2016; Haregeweyn *et al.*, 2013; Linders *et al.*, 2020). In Kenya for instance, the Chamus pastoralists report that *Prosopis juliflora* was introduced twice: first in 1973 through a government initiative; and 10 years later, through the Fuelwood Afforestation Extension Project, a joint initiative from the Food and Agricultural Organization (FAO) and the Government of Kenya (Becker *et al.*, 2016). In Ethiopia, the Afar recall that the species was introduced in the 1980s in state farms and settlements to improve the microclimate, provide shade, halt land degradation, provide fuel wood, as a source of pods for fodder, and to increase sustainability of livelihoods in the Afar region of Ethiopia (Linders *et al.*, 2020). In India, Indigenous Peoples and local communities report that *Prosopis juliflora* was introduced to ameliorate saline soils, and as a source of timber, fuelwood and fibre in the latter half of nineteenth century; and that the species was later promoted by the government from the 1970s onward to combat desertification and soil salinization in North-West India (Duenn *et al.*, 2017).

sequestration (Dickie *et al.*, 2014; Lindenmayer *et al.*, 2012). In fact, a global survey of 226 carbon projects shows that 6 per cent use predominantly alien species and 18 per cent use a mixture of native and alien species (Lindenmayer *et al.*, 2012). Dickie *et al.* (2014) list *Acacia*, *Casuarina* (beefwood), *Eucalyptus, Falcataria* (peacocksplume), *Pinus* (pine) and *Pseudotsuga* (douglas-fir) as major invasive alien genera commonly used for carbon sequestration (**section 3.3.1.1.2**). Similarly, recommendations to introduce alien tree species into British native woodlands as part of adaptive management (**Glossary**) strategies to mitigate rapid climate change, and the potential impacts of associated pests and diseases, will increase the risk of biological invasions as well (Ennos *et al.*, 2019). Overall, although the role of international organizations as indirect drivers facilitating biological invasions has largely been neglected, the evidence to date suggest they may play a role in transport and introduction (*via* trade and infrastructure, see **section 3.2.3.1**) and release and establishment (*via* agriculture, forestry and aquaculture, see **section 3.3.1.1**), which deserves further attention.

Furthermore, international and national programs targeted at biological control of existing pests have resulted in the widespread introduction of invasive alien species. Since 1955, *Euglandina rosea* (rosy predator snail) has been introduced to at least 27 island groups and continental countries, including many Pacific islands, in most cases with the aim of controlling the invasive *Lissachatina fulica* (giant African land snail). The effects in terms of control of the African snail have been limited, but these releases have been catastrophic to many native species; *Euglandina rosea* has caused the extinction of 134 land snail species and the declines of many more species (Gerlach *et al.*, 2021). The introduction of *Gambusia affinis* (western mosquitofish) and *Gambusia holbrooki* (eastern mosquitofish) in most temperate and tropical countries as biological control agents for mosquitoes started in the early 1990s and continues to date. These species are now the most widespread fish in the world, recorded in six continents. Mosquito fish have strong negative effects on freshwater ecosystems and on native fish through predation on juveniles and eggs and/or through competition with species with similar ecological niches (W. E. Walton *et al.*, 2012).

Indigenous Peoples and local communities have also reported that national policies limiting land tenure and access rights can be significant drivers of invasive alien species on their lands (IPBES, 2022b). Indigenous Peoples and local communities will often monitor and manage the numbers of invasive alien species and their impacts on their lands and waters (**Chapter 5**), but their ability to do this is greatly reduced if they do not have access or clear ownership of the lands and waters. Access is indeed crucial for monitoring and management, and land tenure can be essential for communities to actively manage their environments. Moreover, many Indigenous Peoples and local communities actively defend their lands from encroachment by industry and other disruptive influences that can also be drivers for invasive alien species (e.g., deforestation; **section 3.3.1**). Lack of clear land tenure or access rights can also prevent Indigenous Peoples and local communities from effectively defending their lands against this environmental degradation, which can in turn lead to an increase in invasive alien species. Indigenous Peoples and local communities have also noted that lack of access to lands and waters and lack of land tenure can lead to communities leaving.

3.3 THE ROLE OF DIRECT DRIVERS OF CHANGE IN NATURE, NATURAL DRIVERS AND BIODIVERSITY LOSS ON INVASIVE ALIEN SPECIES

This section examines five classes of direct drivers of change in nature that together encompass major human influences on the distribution and abundance of invasive alien species **(Table 3.1)**. The definition and classes of drivers are sourced from the IPBES Global Assessment (IPBES, 2019) but adapted for the purposes of invasive alien species by selecting drivers of relevance for biological invasions. This section thus considers land- and sea-use changes, including farming, fishing, logging (**section 3.3.1**); direct exploitation of natural resources, such as mining and species harvesting (**section 3.3.2**); pollution, including both aerial, soluble and solid waste, with a focus on eutrophication and marine debris (**section 3.3.3**); climate change, including both long-term trends and climatic variability, as well as CO_2 fertilization, changes in climate-related extremes and sea level rise (**section 3.3.4**), and the role of invasive alien species and the management of biological invasions, through biotic facilitation and biological control (**section 3.3.5**). Each subsection first assesses the overall trends in and influence of the driver on invasive alien species, including interlinkages with other drivers, and then, where data allow, notes the specific effects of the drivers on particular biomes, taxonomic groups and units of analysis.

3.3.1 Land- and sea-use change

Land-use change is the major driver of change in nature causing loss of biodiversity and natural habitats globally, affecting close to 75 per cent of ice-free land areas (IPBES, 2019). Agriculture and forestry are major causes of land-use change, for example, global crop production has increased by about 300 per cent since 1970, with crops now occupying half of the habitable land on Earth (IPBES, 2019). Much of the information available for sea-use change is within the context of climate change and specifically changes in physical and biogeochemical properties of the ocean in response to climate warming (IPCC, 2019). However, degradation and loss of natural habitats can occur with many sea-uses, including mining and mineral extraction, coastal developments, land reclamation, wind energy and recreational aquaculture (Vrees, 2021). Some of these sea-uses are anticipated to remain at a constant level but increases are forecast for others. For example in the North Sea, surface mineral extraction, water sport recreation, wind farms and possibly mariculture are all projected to increase (Vrees, 2021). Land- and sea-use change stem from major economic (**section 3.2.3**) and demographic (**section 3.2.2**) indirect drivers.

According to some Indigenous Peoples and local communities, land- and sea-use change is the main driver affecting the establishment and spread of invasive alien species on their lands and seas (**section 3.6.2**, **Box 6.13**). A recent review of native and invasive alien plant, vertebrate and invertebrate biodiversity on islands revealed that number and abundance of alien species are generally higher in areas affected by land-use (plantation forests, agricultural or urban sites) as compared to native habitat (794 alien taxa assessed), whereas the opposite is the case for native biodiversity (5517 native taxa assessed; Sánchez-Ortiz *et al.*, 2020). Habitat fragmentation and agricultural intensification were found to be the most commonly studied drivers facilitating plant invasions in a review by Vilà & Ibáñez (2011), with few studies focusing on the roles of habitat loss, land abandonment and afforestation. Land-use changes, in particular agricultural practices, are an important driver facilitating the spread of fungal and bacterial plant pathogens (Anderson *et al.*, 2004). Studies on alien pathogens tend to highlight their role within agricultural, horticultural or forestry systems, however, with less focus on spread to native systems (Anderson *et al.*, 2004; Panzavolta *et al.*, 2021).

Land- and sea-use change can affect invasive alien species in two main ways, firstly, by directly increasing the rate of introduction of alien species (**section 3.3.1.1**), either intentionally (e.g., through the specific use of alien crops and livestock), or unintentionally (e.g., as contaminants of agricultural or aquacultural commodities). Secondly, a variety of land- and sea-use change related processes increase the vulnerability of native ecosystems to invasive alien species (Vilà & Ibáñez, 2011), including habitat fragmentation (**section 3.3.1.2**), establishment of corridors of disturbed habitat through which alien species can spread (**section 3.3.1.3**), deployment of infrastructure (**section 3.3.1.4**), altering disturbance regimes (**section 3.3.1.5**), or other forms of anthropogenic landscape degradation (**section 3.3.1.6**). **Section 3.3.1** thus describes evidence for links between specific land- and sea-use change drivers and invasive alien species **(Figure 3.15)**, and makes reference made to other indirect and direct drivers when relevant. The demographic and economic indirect drivers behind these land- and sea-use change are described in **section 3.2**.

3.3.1.1 Intentional or unintentional introductions from the use of alien species in terrestrial and marine bioproduction systems

Industries based around the growth and harvest of biological resources, including agriculture, aquaculture, forestry, biofuel and carbon sequestration, forage production and

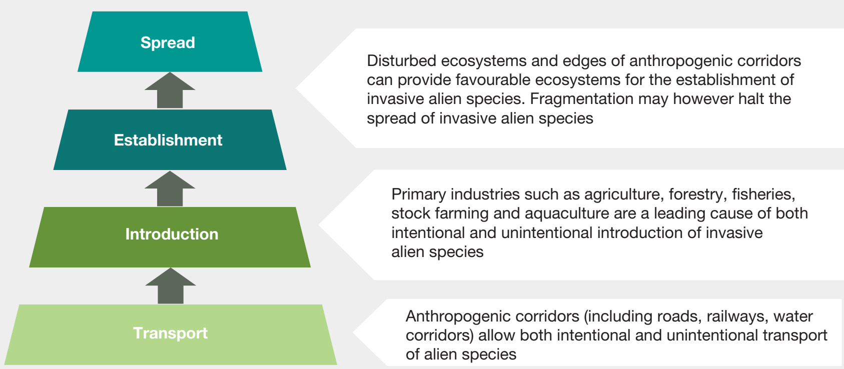


Figure 3 15 **Examples of the role of land- and sea-use change in facilitating invasive alien species across stages of the biological invasion process.**

Anthropogenic corridors can facilitate both the intentional and unintentional transport of invasive alien species, primary industries can facilitate their introduction, and disturbance can facilitate their establishment and spread.

agriculture, are an important cause of the introduction of invasive alien species, both intentionally (e.g., release of plants, animals, or organisms used for pest or weed control into terrestrial or aquatic environments) and unintentionally (e.g., spread of weeds, pathogens, pests and escape from fields/plantations or other containment) (Hulme *et al.*, 2008). Invasive alien species, particularly those from agricultural systems, can facilitate the spread and establishment of pathogens that would otherwise be absent from the introduced range and may pose risks for disease transmission for humans, domestic animals and other native wildlife (Chinchio *et al.*, 2020).

3.3.1.1.1 Aquaculture

Aquaculture (farming of marine organisms) is the largest sector worldwide, after shipping, responsible for introduction of marine alien species (Ojaveer *et al.*, 2018; Bailey *et al.*, 2020). Although the production of marine fish and crustaceans has grown since the millennium, it is now eclipsed by the live-weight volume of marine bivalves and seaweeds which has grown, respectively, from 10 and 11 to 18 and 32 million tonnes per annum between 2000 and 2018 (FAO, 2021). Both the farmed species and/or storage (and other) infrastructures (e.g., cages, nets, floats and ropes) serve as agents of intentional and unintentional introductions and spread of invasive alien species (Campbell *et al.*, 2017). Farmed species are either placed in the sea in enclosures (cages, net pens, rafts), or intentionally released into the environment (Naylor *et al.*, 2001). Intentional introductions comprise both legal and clandestine translocations (Özcan & Galil, 2006; Stentiford & Lightner, 2011; Megahed, 2014). Unintentional introductions consist of “spillover” from crops farmed or stocked in natural habitats (e.g., marine algae farming, marine stock enhancement); escape, unintentional release, or spawning from culture facilities (Arechavala-Lopez *et al.*, 2018, 2017); and alien species associated with the farmed species, or equipment used for culture or transportation of the farmed species (Reise *et al.*, 1998; Mineur *et al.*, 2007). Arguably, due to the high permeability of marine farming facilities, the use of alien species in these settings are essentially intentional introductions to the wild (FAO, 1995; Grosholz *et al.*, 2015). *Tilapia* species are the third most important fish in aquaculture globally, and have established as alien in every country where they have been introduced (Canonico *et al.*, 2005). *Magallana gigas* (Pacific oyster) is one of the most widely used marine invertebrates, introduced primarily for aquaculture in 66 countries, of which alien populations have established in at least 17 (Herbert *et al.*, 2016). The open sea farming of *Undaria pinnatifida* (Asian kelp) and carrageenan-producing seaweeds *Kappaphycus alvarezii* (elkhorn sea moss) and *Eucheuma denticulatum* (eucheuma seaweed), has facilitated their spread to surrounding areas, including marine protected areas, in European, Indian, South America and African coastal waters (Floc’h *et al.*, 1991; Chandrasekaran *et al.*, 2008; Barrios *et al.*, 2007; Tano *et al.*, 2015).

Freshwater aquaculture involves mainly fish (De Silva *et al.*, 2009; Teletchea, 2019) and crayfish (Lodge *et al.*, 2012; Madzivanzira *et al.*, 2021). Globally, the number of alien freshwater fish species is positively correlated with aquaculture production (Gozlan, 2008), and the establishment of alien species from aquaculture has been documented for all continents where introductions have occurred (Britton & Orsi, 2012; De Silva *et al.*, 2006; De Silva, 2012; Gozlan, 2008; Lin *et al.*, 2015; J. Liu & Li, 2010; Luo *et al.*, 2019; Nunes *et al.*, 2015; Ortega *et al.*, 2015; Saba *et al.*, 2020; Shelton & Rothbard, 2006; Q. Wang *et al.*, 2015). Aquaculture is a major source of invasive alien species at national and regional scales. Of the approximately 500 documented introductions of freshwater fishes in the Mediterranean basin, more than 35 per cent are associated with aquaculture (Tricarico, 2012). In the Balkans, 36 species of freshwater fish have been introduced into inland waters *via* aquaculture (Piria *et al.*, 2018). In California, 126 alien species associated with commercial aquaculture have been reported, of which 106 have become established and are negatively affecting native species.

Parasites of a wide range of farmed marine organisms also appear as invasive alien species worldwide. *Haplosporidium nelsoni* (MSX oyster pathogen), a parasite of the Pacific oyster, has spread to estuaries from Maine to Florida, affecting the native *Crassostrea virginica* (eastern oyster; Andrews, 1984; Burreson *et al.*, 2000). Outbreaks of the intrahemocytic parasite *Bonamia ostreae*, protozoan parasite *Marteilia refringens*, two species of parasitic copepods *Mytilicola orientalis* (oyster redworm) and *Myicola ostreae* and the *Ostreid herpesvirus* (OsHV-1), affect farmed Pacific oysters and all originate from imports of stock (Mineur *et al.*, 2014). *Anguillicola crassus*, a blood feeding swimbladder parasitic nematode in eels native to eastern Asia, was widely introduced with its host, *Anguilla japonica* (Japanese eel), to Europe and North America, where it is now widespread in native eel populations sometimes at a prevalence up to 82 per cent of adult and juvenile eels (Barse *et al.*, 2001; Aieta & Oliveira, 2009; T. C. Pratt *et al.*, 2019; Warshafsky *et al.*, 2019). The illegal importation of penaeid prawns from Turkey into Italy was revealed only when the white spot disease was detected (Stentiford & Lightner, 2011). The high concentration of salmon farming in seas off Europe and Canada has been implicated in outbreaks of sea lice (*Lepeophtheirus salmonis* (salmon louse), *Caligus* spp.) in wild salmon (*Oncorhynchus gorbuscha* (pink salmon), *Oncorhynchus keta* (chum salmon)) in these waters (Krkošek *et al.*, 2005). Farmed pacific oysters have served as primary and secondary vectors for the introduction for algae, invertebrates and pathogens affecting both farmed and native oysters (Wolff & Reise, 2002; Mineur *et al.*, 2007, 2014; Verlaque *et al.*, 2015), including invasive alien species such as *Gracilaria vermiculophylla* (black wart weed), *Codium fragile* (dead man's fingers), *Sargassum muticum* (wire weed), *Undaria pinnatifida* (Asian kelp), the sea squirts *Botrylloides violaceus* (violet tunicate), *Didemnum vexillum* (carpet sea squirt) and *Styela clava* (Asian tunicate).

Fish breeds such as farmed *Salmo salar* (Atlantic salmon) have undergone domestication, i.e., intensive selective breeding, and a limited pool of domesticated broodstock, eggs and sperm is shared worldwide (Roberge *et al.*, 2008; Solberg *et al.*, 2020). In the past half century tens of millions of farmed salmon have escaped into the wild (Wringe *et al.*, 2018). Interbreeding between farmed and wild Atlantic salmon in Norwegian waters has altered age and size at maturation in 62 wild salmon populations, caused widespread changes to fitness-related life-history traits, thus threatening already vulnerable wild populations (Karlsson *et al.*, 2016; Bolstad *et al.*, 2017). Interbreeding between farmed and wild conspecific populations is also found in *Sparus aurata* (gilthead sea bream), *Dicentrarchus labrax* (European sea bass), *Argyrosomus regius* (brown meagre) and *Gadus morhua* (Atlantic cod) (Somarakis *et al.*, 2013; Jørstad *et al.*, 2014; Izquierdo-Gómez *et al.*, 2017; Arechavala-Lopez *et al.*, 2017, 2018). *Symphodus melops* (corkwing wrasse), used to control sea lice in salmon farms, also escapes and hybridizes with individuals in local populations (Faust *et al.*, 2018).

3.3.1.1.2 Forestry, agroforestry, biofuel and carbon sequestration

Trees and shrubs have been introduced globally for wood production, fruit and seed crops, erosion control, live fences and building material (Richardson, 1998; Richardson & Rejmánek, 2011; van Kleunen *et al.*, 2020). Of introductions of invasive alien trees and shrubs, 13 per cent were attributed to forestry, 10 per cent to food, and seven per cent to agroforestry (Richardson & Rejmánek, 2011). Many trees extensively used in forestry have high potential to become invasive alien species, including the genera *Pinus* (Richardson, 2006; Fernandes *et al.*, 2016; Brundu *et al.*, 2020), *Acacia* (Donaldson *et al.*, 2014; Richardson *et al.*, 2011, 2015) and *Eucalyptus* (Bennett, 2010; Forsyth *et al.*, 2004; Hirsch *et al.*, 2020; Simberloff & Rejmanek, 2011). Of the roughly 100 *Pinus* species, at least 17 species are now considered as invasive alien species in natural ecosystems, particularly in the southern hemisphere (Richardson & Blanchard, 2011; Richardson & Nsikani, 2021). Of the approximately 200 *Eucalyptus* species cultivated within South Africa (Henderson, 2009), six are listed as invasive alien species by the National Environmental Management Biodiversity Act. Studies in the Iberian Peninsula also show *Eucalyptus globulus* (Tasmanian blue gum) can spread from plantations (Fernandes *et al.*, 2016). *Acacia* species from Australia are widely distributed invasive alien species (Le Maitre *et al.*, 2011). Over 70 *Acacia* species were introduced throughout South Africa for forestry, dune stabilization and ornamental use during the 19th and 20th centuries (Bennett, 2011). Of these 70 introduced species, 14 are now considered to be invasive alien species, four of which arose from commercial forestry plantations (Van Wilgen *et al.*, 2011). The southern hemisphere (e.g., South America, Oceania) has been particularly affected by tree invasions, because of the massive scale of commercial plantations and the absence of competition by native tree species (García *et al.*, 2018; Nuñez *et al.*, 2017; Richardson *et al.*, 2021).

Increased interest in biomass-based energy has increased the use of alien species with rapid growth rates, ease of establishment, wide environmental tolerances, and prolific seed production in plantations, characteristics that also promote them as potential invasive alien species (Barney & DiTomaso, 2008; Leakey, 2009; Richardson & Blanchard, 2011). Across the south-eastern United States, *Eucalyptus* species are commonly utilized for bio-energy (Callaham *et al.*, 2013; Lorentz & Minogue, 2015), and have the potential to invade surroundings woodlands (Callaham *et al.*, 2013). Large stature grasses, such as bamboo, are also used as common biofuel crops, with "running" bamboo species noted to present a significantly higher risk of biological invasion than "clumping" species (Lieurance *et al.*, 2018). The perennial grasses *Miscanthus sinensis* (eulalia) and *Miscanthus sacchariflorus* (Amur silvergrass), planted for ornamental and biofuel uses, also pose high risks across Europe and North America, especially in grassland and tall herb vegetation, ruderal habitats and roadsides (Schnitzler & Essl, 2015). Climate modelling indicates *Miscanthus* (silvergrass) species have already been introduced to most of the suitable regions in the northern hemisphere, whereas there is climatic potential for further expansion in the southern hemisphere, suggesting increased future biological invasion threat there (Hager *et al.*, 2014).

Food forestry, whereby a diversity of food plants is planted in natural or seminatural forested ecosystems, may represent an emerging pathway for the introduction, establishment and spread of alien species into natural or near-natural ecosystems. For example, a recent study identified almost 500 alien species used in the fast-growing food forestry sector in the Netherlands alone, including a number of high -risk invasive aliens including *Akebia quinata* (five-leaf akebia), *Helianthus tuberosus* (Jerusalem artichoke), *Rhus typhina* (staghorn sumac), *Rosa rugosa* (rugosa rose) and *Vaccinium macrocarpon* (cranberry) (Hoppenreijs *et al.*, 2019).

Plantation forests are also hotspots (invasion hotspot in the **Glossary**) for unintentional introductions of invasive alien species In a global review of invasive alien plants, vertebrates and invertebrates on islands (794 alien species), plantation forests had consistently higher numbers and abundances of alien species as compared to native habitat (Sánchez-Ortiz *et al.*, 2020). Invasive forest pathogens have been responsible for many disease outbreaks across commercial, natural and urban forest ecosystems, and generally occur as a result of unintentional introductions *via* containment or stowaway (Burgess *et al.*, 2016; Migliorini *et al.*, 2015; Paap *et al.*, 2020). Notable examples include; *Cryphonectria parasitica* (blight of chestnut), *Ophiostoma novo-ulmi* (Dutch elm disease), *Phytophthora cinnamomi* (Phytophthora dieback), *Phytophthora ramorum* (sudden oak death) and *Hymenoscyphus fraxineus* (ash dieback) (Brasier & Buck, 2001; Pautasso *et al.*, 2013; Rigling & Prospero, 2018; Rizzo & Garbelotto, 2003; Shearer *et al.*, 2007).

3.3.1.1.3 Forage production and livestock grazing

Forage production and pastures for domestic herbivores is a major land-use in almost all biomes of the world (Brondizio *et al.*, 2019). Management of such lands is a major source of biological introductions, because species sown or planted for forage, or weeds associated with these land-uses, may escape and spread into natural ecosystems (Nuñez *et al.*, 2017; O'Connor & van Wilgen, 2020; Pándi *et al.*, 2014). In a survey by Driscoll *et al.* (2014), 91 per cent of grasses developed by agribusiness for pasture were listed as weeds somewhere in the world, and often in the same countries where they were actively been developed and marketed. Policies aiming to facilitate pastoral development can also be responsible for introductions, such as occurred for the pasture grass *Andropogon gayanus* (tambuki grass) in Australia (Cook & Dias, 2006), which is now known to increase wildfire intensity, and transform species-rich savannah systems into alien-dominated grasslands (Driscoll *et al.*, 2014; **section 3.2.5**). In Texas, United States, alien grass species were seen as the future of forage production during the early twentieth century, and several introduced species have since escaped pasture cultivation: *Bothriochloa ischaemum* (yellow bluestem), *Dichanthium annulatum* (Kleberg's bluestem), *Dichanthium aristatum* (angelton bluestem), *Cenchrus ciliaris* (buffel grass), *Megathyrsus maximus* (Guinea grass), *Eragrostis lehmanniana* (Lehmann lovegrass) and *Cynodon dactylon* (Bermuda grass) (Wied *et al.*, 2020). Several of these economically important pasture grasses are invasive alien species throughout several countries. *Megathyrsus maximus*, with its high yield, palatability and tolerance of herbivory is now considered a weed species throughout Africa, America, Australia and Asia (Randall, 2017; Rhodes *et al.*, 2021). *Cenchrus ciliaris* has become a problematic species across Australia, the United States, Mexico and South America (V. M. Marshall *et al.*, 2012) and Bermuda grass now also has a cosmopolitan distribution and is considered one of the world's worst weeds (Randall, 2017; Way, 2014). Some invasive alien species in agricultural systems were introduced through planting of windbreaks and hedgerows, including the globally versatile *Ulex europaeus* (gorse), one of the most invasive alien species in the world, introduced from Europe to Australia, Chile, New Zealand, Sri Lanka and the United States (Roberts & Florentine, 2021). In Southern Africa, invasive alien *Opuntia* (pricklypear) species were initially grown as wind-breaks, fences, and also supplementary fodder sources (S. E. Shackleton & Shackleton, 2018).

Some introduced species also act as hosts of further invasive alien species. For example, in the United States, *Festuca arundinacea* (tall fescue), a cool season-grass introduced from Europe, dominates grasslands and is considered an invasive alien species across multiple states (Barnes *et al.*, 2013; Pfeifer-Meister *et al.*, 2008). The spread of tall fescue is concerning as it also may act as a reservoir host for *Alternaria* (fungal pathogen), which produces crop damaging mycotoxins (H. E. Wilson *et al.*, 2014).

3.3.1.1.4 Agriculture

In a global study of terrestrial plant invasions 407 (46 per cent) of 886 alien plants were introduced intentionally through agricultural pathways (Turbelin *et al.*, 2017). This study reports three of the top five terrestrial invasive alien plant species globally to have their main introduction pathways associated with agricultural practices: *Cyperus rotundus* (purple nutsedge; found in 37 per cent of countries), *Ricinus communis* (castor bean; 31 per cent) and *Leucaena leucocephala* (leucaena; 27 per cent). Agricultural use is also a major source of aquatic plant invasions. In China, several alien freshwater aquatic plants have been introduced for landscaping, water purification and forage purposes, five of which are now considered to be invasive alien species: *Sporobolus alterniflorus* (smooth cordgrass), *Azolla filiculoides* (water fern), *Alternanthera philoxeroides* (alligator weed), *Urochloa brizantha* (palisadegrass) and *Urochloa mutica* (para grass) (Wu & Ding, 2019).

The occurrence of some ungulates as invasive alien species arose from agricultural practices (Spear & Chown, 2009). Farms or hunting of species for fur has resulted in the intentional introduction, or escape from farms or captivity, of: *Mustela vison* (American mink) in Europe (Bonesi & Palazon, 2007; E. J. Fraser *et al.*, 2017); *Procyon lotor* (racoon) in Europe (Beltrán-Beck *et al.*, 2012); *Oryctolagus cuniculus* (rabbits) throughout the world (Lees & Bell, 2008) and particularly in New Zealand (C. M. King & Forsyth, 2021), Australia (Myers *et al.*, 1994) and South America (Howard & Amaya, 1975; Iriarte *et al.*, 2005); *Vulpes vulpes* (red fox) in Australia (Saunders *et al.*, 2010); and *Trichosurus vulpecula* (brushtail possum) in New Zealand (Clout, 2006; C. Jones *et al.*, 2012). **Section 3.3.2.1** on stocking for hunting and **section 3.2.3.3** on pet trade also discuss how these drivers have facilitated invasive alien species.

Agriculture has also facilitated biological invasions by plant and animal pathogens, parasites and diseases. Alien plant pathogens may be introduced *via* seeds and soil used in agriculture (Pimentel *et al.*, 2001). For example, 65 per cent of plant pathogens in the United States were considered alien species (Pimentel *et al.*, 1992), 74 per cent in the United Kingdom (Carlile, 1988), 82 per cent in Australia (Persley & Syme, 1990), 85 per cent in South Africa (Nel, 1983), 74 per cent in India (Singh, 1985), and 75 per cent in Brazil (Echandi *et al.*, 1972). Many microbes and other parasites accompany livestock as they are introduced into new countries (Pimentel *et al.*, 2001). *Mycobacterium bovis* (bovine tuberculosis) was introduced to many places, including New Zealand, as cattle were transported out of Europe during the nineteenth century (N. H. Smith, 2012). In New Zealand, bovine tuberculosis is now also prevalent in other invasive alien species including *Trichosurus vulpecula* (brushtail possum), *Sus scrofa* (feral pig), *Mustela putorius furo* (ferret), *Mustela erminea* (ermine), *Erinaceus europaeus* (European hedgehog) and deer species (Livingstone *et al.*, 2015). In Europe, the introduction of *Procyon lotor* (racoon) also established *Baylisascaris procyonis* (raccoon roundworm), which may potentially induce central nervous system disease in humans (Chinchio *et al.*, 2020).

3.3.1.2 Fragmentation of ecosystems

Increasing exploitation of natural resources and land-use changes have led to widespread fragmentation of terrestrial ecosystems, so that 70 per cent of remaining forest areas globally are now within 1 km distance of a forest edge (IPBES, 2019). Fragmentation is usually associated with loss of total habitat area, changes in habitat quality, and increased biotic and abiotic influence from the surrounding landscape (Eriksson *et al.*, 2002; Sodhi *et al.*, 2010; Vilà & Ibáñez, 2011). Fragmentation of landscapes and habitats is one of the most significant process driving decrease of native biodiversity and species richness globally (IPBES, 2019; Millennium Ecosystem Assessment, 2005b).

Fragmentation increases the proportion of the native habitat exposed to edge effects, where higher propagule pressure and faster growth of pioneer and generalist species, many of which are alien, can drive replacement of native habitat specialists (Laurance & Peres, 2006; Lobo *et al.*, 2011; B. A. Santos *et al.*, 2008; Tabarelli *et al.*, 2008). The increased edge-to-interior ratio of fragmented landscapes increases the prevalence of invasive alien species in fragments, as shown for plant and lepidopteran diversity in South Texas (Stilley & Gabler, 2021), woody plant diversity in New England (J. M. Allen *et al.*, 2013), *Hovenia dulcis* (Japanese raisin tree) in Brazilian Atlantic forest patches (Padilha *et al.*, 2015) and *Sporobolus alterniflorus* (smooth cordgrass) in mangroves along the coast of China (Z. Zhang *et al.*, 2021). Recent studies also document increased spread of invasive alien species in fragmented landscapes over time, such as the study from Achury *et al.* (2021) of *Linepithema humile* (Argentine ant) invading coastal southern California. Similarly, *Ulex europaeus* (gorse) spread widely in fragmented landscapes of south-central Chile while large intact forest areas experienced lower rates of invasion over the same time period (Altamirano *et al.*, 2016).

As fragmentation increases, the remaining fragments are more isolated from each other, which may both promote and hinder biological invasions. Increased patch isolation may promote biological invasions if invasive alien species are more common in the habitats surrounding the patches, as has been shown for alien pasture grasses which frequently occur in Australian forest fragments surrounded by landscapes with high pasture cover (S. Butler *et al.*, 2014). Another example is the prevalence of *Aulacaspis yasumatsui* (cycad aulacaspis scale) on cycads in Guam, where isolated fragments suffered greater damage from these alien scale insects than did fragments with higher connectivity (Marler & Krishnapillai, 2020). Connectivity of native habitats may also be promoting the spread of invasive alien species which are dependent upon native dispersal vector species that depend on this habitat (e.g., Guiden *et al.*, 2015).

Some cases exist in which the fragmented habitat is less favourable to invasive alien species. Insect pests and specialist pathogens of forest trees are less common in counties of the United States which have more fragmented forests (Guo, Riitters, *et al.*, 2018). In the case of animals, patches further apart than the organism is able to cross may hinder the spread of invasive alien species (e.g., Bridgman *et al.*, 2012). The reproductive success of the invasive alien tree *Ligustrum lucidum* (broad-leaf privet) is lower in forest fragments than continuous forests, not due to lower seed production but due to unfavourable soil conditions for seedling establishment in fragments (Aguirre-Acosta *et al.*, 2014). In aquatic environments, artificial fragmentation (e.g., underwater barriers) may slow down the spread of alien species, even though some barriers may be more effective for native species than for alien species (Airoldi *et al.*, 2015).

Fragmentation of native habitat also creates corridors for invasive alien species (**section 3.3.1.3**), increased disturbance (**section 3.3.1.5**) and lower patch habitat quality (**section 3.3.1.6**). A mechanism by which fragmentation promotes biological invasion may be through rendering native populations more vulnerable to local extinction (Hanski, 1999), leaving vacant niches and hence decreasing biotic resistance to invasions and ecosystem resilience (**section 3.4.2**). Most of the evidence for the effects of fragmentation on biological invasions comes from invasive alien plants or plant pest species, and specifically in relation to the spread stage of the biological invasion process (**Chapter 1**, **section 1.4.4**).

3.3.1.3 Creation of anthropogenic corridors

Different types of anthropogenic corridors act as major routes for the transport and spread of invasive alien species (e.g., Galil *et al.*, 2015; Hulme *et al.*, 2008), although these are often not explicitly considered in pathway assessments (CBD, 2014; Leclerc *et al.*, 2018). Anthropogenic corridors, including roads, highways, railways, hiking trails, tunnels, pipelines, power lines, canals and bridges, are rapidly expanding for trade, travel and transport (**sections 3.2.3.1** to **3.2.3.4**). It is projected that length of roads will increase by over 60 per cent (or to between 3 and 4.7 million km) globally from 2010 to 2050 (Dulac, 2013), a large fraction of which is projected to be built in developing countries in some of the world's last remaining wilderness areas, such as the Amazon, the Congo basin and New Guinea (Meijer *et al.*, 2018). The volume of freight transported *via* anthropogenic corridors has consistently grown since the 1960s especially in Europe and North America (**section 3.1.1**; Hulme, 2009a).

Anthropogenic corridors allow both intentional and unintentional transport of invasive alien species, and they create disturbed and transformed habitat such as road and canal verges that allow subsequent establishment and spread of invasive alien species into otherwise impassable regions. The mechanism of the influence of anthropogenic corridors on the stages of the biological invasion process can be summarized as follows; 1) allowing easier transport and spread of invasive alien species by natural (e.g., wind, water, animals) or human mediated (e.g., cars, trains, ships, people) vectors, 2) facilitating establishment of invasive alien species by disturbing, stressing or removing native species and ecosystems along corridor verges, 3) providing new corridor verge habitats for invasive alien species to establish and spread by altering abiotic environmental conditions (e.g., soils, hydrology, wind; Trombulak & Frissell, 2000).

In terrestrial biomes, several global surveys provide evidence that the abundance and diversity of alien plants is higher along roads compared to adjacent native habitat, and decreases with distance away from the roads (Lázaro-Lobo & Ervin, 2019; Suárez-Esteban *et al.*, 2016). The long linear features of roads and railways facilitate the long-distance dispersal of alien seeds (Hulme, 2006). In a selectively logged tropical forest in Bolivia, logging vehicles spread the seeds of the alien *Megathyrsus maximus* (Guinea grass) at least 500 m from the established populations (Veldman & Putz, 2010). Road density and road age also positively correlate with alien species' distributions (Hulme, 2009). For example, alien earthworms have spread farther from older roads in boreal forests of Canada (Cameron & Bayne, 2009), suggesting roads provide fringe sources for colonization of native habitat. Anthropogenic corridors also drastically alter the surrounding biotic, physical and chemical environments. The edges of anthropogenic corridors (e.g., roadsides, highways, railways) provide favourable habitats for the establishment of alien plants (e.g., M. J. Hansen & Clevenger, 2005; Jodoin *et al.*, 2008; Sullivan *et al.*, 2009). Rural roads, mountainside highways and powerlines change the surrounding plant species composition, and enhance the establishment of invasive alien species in mountainous and protected areas (L. G. Anderson *et al.*, 2015; Mortensen *et al.*, 2009; Rentch *et al.*, 2005; Spooner, 2015; Wagner *et al.*, 2014). Some Indigenous Peoples and local communities also observe the role of anthropogenic corridors in facilitating biological invasions, for instance, people in Arunachal Himalayas in India view road construction and road use as the drivers facilitating biological invasion, as well as introduction of cattle which brought alien seeds (Kosaka *et al.*, 2010). In alpine and Arctic ecosystems, the establishment and subsequent spread of alien plants is increased along mountain roads, hiking trails and buried oil pipelines (Alexander *et al.*, 2016; Langor *et al.*, 2014; Liedtke *et al.*, 2020; **section 3.3.5.1**, **Box 3.10**).

In freshwater and marine biomes, canals facilitate the transport, introduction and spread of invasive alien species at global and regional scales (Asth *et al.*, 2021; Bij de Vaate *et al.*, 2002; Boudouresque & Verlaque, 2012). The role of these water corridors in facilitating biological invasions are well studied for fish and aquatic invertebrates (Devin *et al.*, 2005; Karatayev *et al.*, 2008; Rakauskas *et al.*, 2016), especially in Europe (e.g., Katsanevakis *et al.*, 2013). For example, 507 marine alien species have arrived in European Seas through canals, such as the Suez Canal (**Box 3.7**; Katsanevakis *et al.*, 2013). Since the eighteenth century, the connection of the European seas and rivers to the Eurasian waterways *via* canals showed a stepwise increase, and the extensive network of inland waterways has allowed the biological invasion of aquatic alien species from different biogeographical regions (Leuven *et al.*, 2009), such as the establishment of Ponto-Caspian invertebrates throughout the central European corridors (Karatayev *et al.*, 2008). Another example is the shipping canals near Chicago, Illinois that link the Great Lakes with the Mississippi River, which have allowed the exchange of 15 species of fish and invertebrates formerly confined to just one of the basins (Rahel, 2002). Additionally, inter-basin water transfers provide a direct link between previously isolated catchments and thereby modify the water flow, chemistry and temperature of receiving waters. Water inflow *via* canals can result in eutrophication and changes in salinity, thereby allowing the establishment of invasive alien species (e.g., Pienimäki & Leppäkoski, 2004; Sarà *et al.*, 2018).

Thus, creation of anthropogenic corridors is an important driver across all stages of the biological invasion process (transport, introduction, establishment and spread) for various taxa (e.g., plants, vertebrates and invertebrates) in terrestrial, freshwater and marine biomes.

Box 3 7 **The Suez Canal and invasive alien species.**

The Suez Canal, a linchpin of transportation networks between Europe and Asia, carries over 10 per cent of global trade, with19,000 vessels transiting the Canal in 2020 (Veiga, 2021). The Canal is also the main pathway of alien species introduction into the Mediterranean Sea. The Suez Canal was opened over 150 years ago, yet Erythraean species are newly recorded in the Mediterranean to this very day **(Figure 3.16)**. Biological invasions by Erythraean species are driven by the region's environmental characteristics and anthropogenic activities. The latter include: physical changes to the Canal that have impacted its hydrography and hydrology, and increased its potential as a "corridor"; and changes to the Levantine marine environment that have made it more susceptible to biological invasion by modifying its hydrological properties and species diversity, and destabilizing the shelf community structure (Galil, 2006).

The Suez Canal (8 m deep, with cross-section area 304 m^2, when built in 1869) is hydrographically complex, passing through five anthropogenic lakes of widely diverse salinity (Menzalah, Ballah, Timsah, Large and Small Bitter Lakes). The dissolution of the Bitter Lakes' salt bed, complete by the 1960s, removed the early Canal's salinity barrier. A recent study of the Canal's flow intensity and direction (1923–2016) supports unidirectional biological invasions into the Mediterranean, with significant increase in northward flow during the early 1980s following a major expansion (depth from 15.5 to 19.5 m, doubling cross-section area), and a second expansion in 2015 after the opening of the "new" Suez Canal (Biton, 2020). A larger Canal accommodates transit of more and larger vessels, many in ballast and befouled, and discharges a larger volume of Red Sea waters with their entrained biota into the Mediterranean Sea.

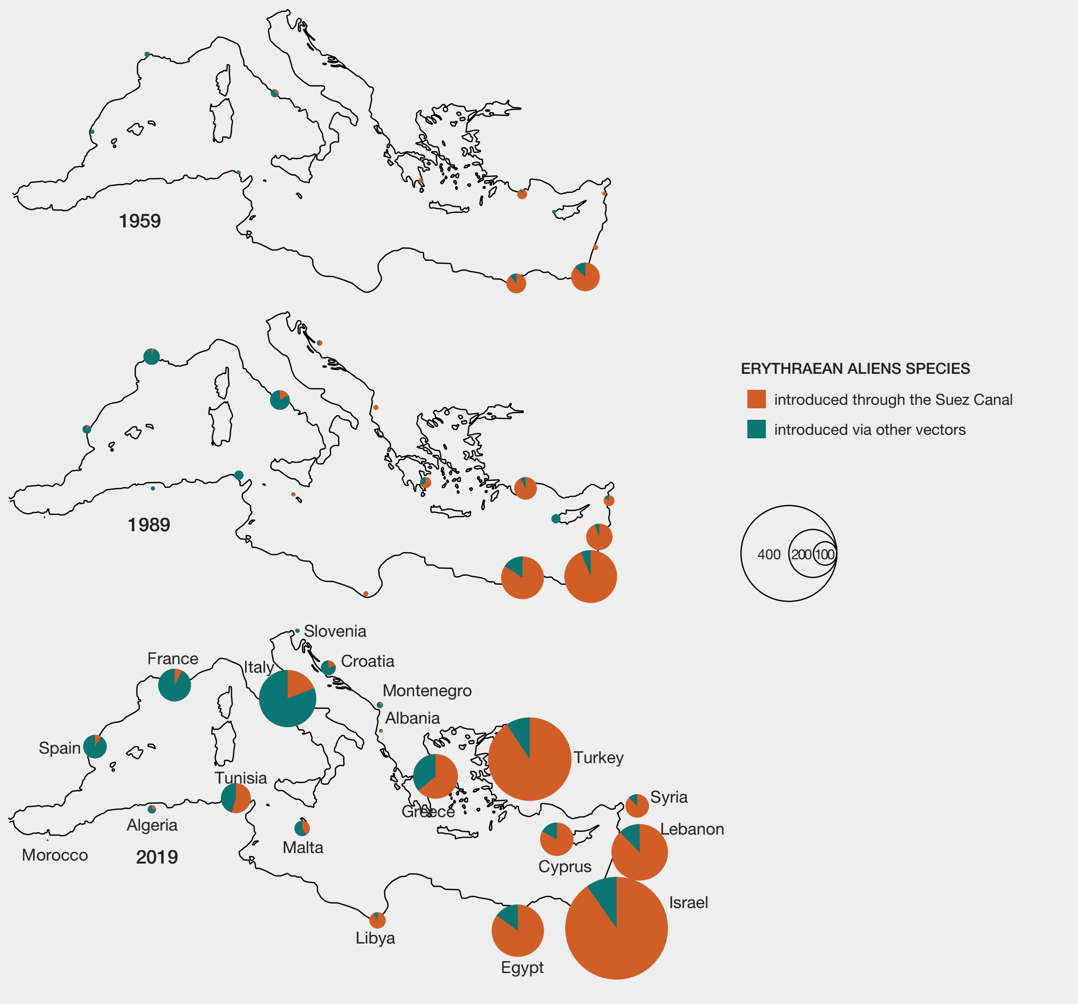


Figure 3 16 **Number of multicellular marine alien species in peri-Mediterranean countries, and their means of introduction, 1959, 1989, 2019.**

Lighter tone: Erythraean aliens (i.e., introduced through the Suez Canal), darker tone: alien species introduced *via* other vectors. A data management report for this figure is available at https://doi.org/10.5281/zenodo.7861139

3.3.1.4 Deployment of marine infrastructure

Coastal landscapes are being transformed through marine urban sprawl with an increase in construction of artificial structures to support commercial, residential and tourist activities (Dafforn *et al.*, 2015; Bulleri & Chapman, 2010). Indeed, more than 50 per cent of the shorelines of some regions of Europe, United States, Australia, and Asia are modified by hard engineering. Offshore aquaculture facilities and offshore energy infrastructures are also increasing in prevalence. Marine urbanization is predicted to escalate in the future as sea-level rises and extreme climate events, including storms, increase in frequency (Dafforn *et al.*, 2015). Artificial structures alter seascapes and the functioning of marine ecosystems through local and regional effects (Bulleri & Chapman, 2010; Todd *et al.*, 2019) including the establishment and spread of invasive alien species.

The potential for invasive alien species to utilize marine infrastructure is widely recognized (Bulleri & Chapman, 2010; Bulleri & Airoldi, 2005; Vaselli *et al.*, 2008). Marine infrastructure can facilitate invasive alien species by providing artificial hard substrates that some invasive alien species can colonize (Farr *et al.*, 2021). Indeed, artificial hard structures such as breakwaters, jetties, seawalls, floating pontoons and pier pilings provide suitable habitats for alien species (Bulleri & Airoldi, 2005; S. L. Williams & Smith, 2007) and can also function as corridors through unsuitable habitats (Bulleri & Airoldi, 2005). Nearshore infrastructure is considered to provide entry points for invasive alien species, with the numbers of invasive alien species on pontoons and pilings being 1.5-2.5 times higher than on natural rocky reefs (Glasby *et al.*, 2007). *Codium fragile* (dead man's fingers), an invasive alien seaweed species native to east Asia, colonized hard structures installed to provide coastal protection in the northern Adriatic Sea and is now found on temperate rocky shores around the world (Bulleri & Airoldi, 2005), and has also replaced native kelp on the leeward shores of the United States (Levin *et al.*, 2002). Similarly, a study on invasive alien ascidians demonstrated that ascidian species spread onto natural habitats from marine infrastructure, and that species differ in the rate and success of this secondary spread (Simkanin *et al.*, 2012). In a global literature survey, the alien ascidians *Botrylloides violaceus* (violet tunicate) and *Botryllus schlosseri* (star ascidian) were reported four times as often in anthropogenic marine habitats relative to natural habitats, while two other alien ascidians, *Didemnum vexillum* (carpet sea squirt) and *Styela clava* (Asian tunicate), were encountered on floating docks, pilings and aquaculture installations eight times as often as they were found in nearby natural habitats (Simkanin *et al.*, 2012). These findings illustrate the differences in biological invasion potential and/or rate between closely related species.

Offshore floating structures, such as wind facilities, can provide substrate for introduced hard-substrate benthic organisms to colonise, and thus can contribute to the further spread of invasive alien species, especially in the intertidal zone (Kerckhof *et al.*, 2016). Deepwater and offshore floating infrastructures generally are considered less likely to be colonized by invasive alien species than nearshore infrastructures (Farr *et al.*, 2021) because the nearshore is often associated with higher human activity and consequently increased pathways of introduction in comparison to offshore locations.

3.3.1.5 Changes in landscape-seascape disturbance regimes (intensification and reduction)

Changes in landscape-seascape disturbance regimes, including both intensification and reduction in disturbance intensity, have been ubiquitous in natural ecosystems as a result of human activities; for example over 50 per cent of the global land area has experienced changed fire regimes, and fires are expected to become more common in coming decades as a result of climate change and increasing human occupation (IPBES, 2018b). Such changes may affect the capacity of invasive alien species to establish and thrive through direct effects of changes in disturbances. Effects may also be more indirect, through interfering with native competitors, grazers, or predators, or through the modification of fire frequencies or nutrient and water regimes (**sections 3.3.4.5**, **3.3.3.1** and **3.3.1.5** respectively). At broader spatial scales, changes in landscape-seascape disturbance regimes may affect invasive alien species through fragmenting landscapes, creating corridors, the deployment of infrastructure, or degrading habitats, but also through protecting areas from disturbance, for example in designated protected areas (**sections 3.3.1.2** to **3.3.1.5**).

In an observational study spanning 200 sites around the world, disturbance *per se* was found to be a weak predictor of plant invasions (Moles *et al.*, 2012). An older literature survey (Lozon & MacIsaac, 1997) found a greater importance of disturbance for successful plant invasions (implicated in 67 per cent of disturbance caused by invasive alien species) than by animals (implicated in 28 per cent of disturbance caused by invasive alien species), particularly during the establishment phase (86 per cent vs. 12 per cent relied on disturbance for plants vs. animals). The role of landscape disturbance in influencing microbial invasive alien species is far less understood than for macro-organisms, but the literature that exists suggests patterns remain similar with sites of high disturbance, anthropogenic impact, fluctuating resource supplies and release from predators resulting in increased establishment of invasive alien microbial species (Litchman, 2010). Overall, increased landscape disturbance *per se* hence seems to have a weak but positive role in facilitating invasions.

A more nuanced analysis focusing on changes in disturbance regimes, however, can provide better predictive power than disturbance regime *per se*. Several reviews have found that changes in land-use regimes, and in particular in human-mediated disturbance regimes related to fire, grazing and agriculture, facilitate plant invasions through both direct and indirect pathways (Jauni *et al.*, 2015; Moore, 2005; Vilà & Ibáñez, 2011). Experimental studies support these observations, frequently indicating that changes to disturbance regimes, both increases and decreases in disturbance frequency and intensity relative to natural or historic levels, and in particular the introduction of novel disturbance types, provide opportunities for alien plant species to establish (Kempel *et al.*, 2013). **Sections 3.3.1.5.1**, **3.3.1.5.2** and **3.3.1.5.3** therefore summarize the ways in which changes, and particularly intensification of human disturbance regimes, increase the establishment and spread of invasive alien species. There is stronger evidence for disturbance effects on terrestrial systems and plants than other systems and taxonomic groups, but some evidence also exists from aquatic and marine systems and for vertebrates, invertebrates and microorganisms. All IPBES regions are represented, with much of the evidence from Australia and the United States. The literature on disturbance regimes as a driver that facilitates invasive alien species covers all stages of the biological invasion process, but with more evidence on the establishment and spread stages, the latter often associated with documentation of the impact on native biodiversity and human livelihoods (**Chapter 4**).

3.3.1.5.1 Agricultural disturbance regimes

In terrestrial biomes, a global meta-analysis (Jauni *et al.*, 2015) found that plant invasions can be facilitated by discrete disturbance events such as fire, agricultural activity, and more generally broad shifts in anthropogenic activity. Specifically, they found that increased domestic grazer activity and general anthropogenic disturbance events increased both the diversity and abundance of alien species, whilst fire- or soil-based disturbance activities led only to increases in diversity of alien species. This trend was stronger in forest ecosystems than in wetlands and grasslands. Importantly, time elapsed since the disturbance occurred was considered critical, with significant responses only observed in cases where communities were monitored more than five years post-disturbance. In line with these findings, in Ghana, conversion to maize production resulted in increasing removal of fallow trees, which has encouraged land degradation and facilitated establishment and spread of *Chromolaena odorata* (Siam weed; Amanor, 1991). In arid and semi-arid rangelands of the United States, livestock overgrazing decreases the plant cover of native palatable grasses and accelerates the dominance of alien annual grasses (e.g., *Bromus tectorum* (downy brome); Chambers *et al.*, 2007; Keeley *et al.*, 2003). In the grasslands of the Austral Andean Mountains in Argentina, establishment of the invasive alien *Pinus halepensis* (Aleppo pine) was considerably higher in areas grazed by feral horses, where perennial grasses were negatively affected by defoliation, giving advantage to the alien plant *Echium plantagineum* (Paterson's curse) (de Villalobos & Schwerdt, 2020; de Villalobos & Zalba, 2010). In Australia, overgrazing by introduced feral camels, buffalo and pigs has facilitated invasive alien species establishment in arid and semi-arid ecosystems throughout the country (Burrows, 2018; Sloane *et al.*, 2021). Invasive alien plants often have higher performance and higher resource use efficiency than coexisting native species, suggesting a higher ability to benefit from increased resource availability resulting from changes in disturbance regimes as a potential mechanism (e.g., Daehler, 2003; Kolar & Lodge, 2001; Leishman & Thomson, 2005). Among animals, *Solenopsis invicta* (red imported fire ant) was characterized as a "disturbance specialist" when subjected to mowing and ploughing regimes, to the extent that the species was found not to invade forest habitats of native ants in the absence of such disturbances (J. R. King & Tschinkel, 2008). Increasing agricultural disturbance also benefited alien predatory *Coccinellidae* (ladybeetles) in Chile, and these alien ladybeetles could be considered "disturbance specialists" (Grez *et al.*, 2013). Modified landscapes may also support higher abundances of invasive alien animals than unmodified landscapes, for example introduced *Vulpes vulpes* (red fox) thrive within Australian agricultural systems (Graham *et al.*, 2012; Towerton *et al.*, 2011).

Biological invasions in terrestrial systems may be affected by land abandonment as well as land-use intensification. After abandonment, the succession from agricultural to forested landscapes is generally associated with an increased spread in plant invasions, particularly after crop abandonment (Vilà & Ibáñez, 2011), which may be linked to higher competitive ability and hence establishment and spread of some invasive alien species under the reduced disturbance intensity and frequency in post-abandonment vegetation (van der Zanden *et al.*, 2017). The role of abandonment in facilitating invasive alien species is well known among Indigenous Peoples and local communities in several parts of the world. In the Panchase area of Nepal, Indigenous communities report that land-abandonment has led to the establishment and spread of invasive alien species (Schwilch *et al.*, 2017). In the Amatole District of the Eastern Cape, South Africa, local communities noticed that the abandonment of arable fields coupled with the dispersal of seeds by local birds have led to invasion by *Lantana camara* (lantana; Jevon & Shackleton, 2015). Indigenous Peoples and local communities in South Africa have also observed that some abandoned agricultural lands can become hotspots of invasive alien species (C. M. Shackleton & Gambiza, 2008).

3.3.1.5.2 Changes to fire regimes

Fire is a key natural disturbance process that plays an important role in regulating community composition and ecosystem functioning in a diversity of ecological systems worldwide (He *et al.*, 2019; Keeley *et al.*, 2011; Pausas & Keeley, 2019). In fire-adapted systems, continuation of the historic natural or anthropogenic fire regimes generally has relatively little influence on either native species performance or the establishment of alien species whereas loss of traditional fire regimes may benefit invasive alien species (Alba *et al.*, 2015; L. T. Kelly *et al.*, 2020; Velle *et al.*, 2014). A meta-analysis of the role of fire on native and invasive alien species found that prescribed low-intensity burns may benefit native species but generally do not affect alien species, whereas wildfires consistently enhanced alien species performance and diversity, and especially so in arid shrublands, temperate forests and heathlands (Alba *et al.*, 2015). The evidence from this global study was largely from the United States and Australia, pointing to regional knowledge gaps. Invasive alien species may benefit from both reduction and enhancement of fire regimes relative to historic levels. For example, in naturally fire-prone prairies in the United States, fire suppression led to the dominance of invasive alien earthworms (Callaham *et al.*, 2003; Callaham & Blair, 1999), and increased frequency or intensity of fire relative to historic fire regimes tended to increase diversity and performance of alien plants, whereas native biodiversity was highest when historic fire regimes were maintained (D'Antonio, 2000). In tussock grasslands in New Zealand, alien spiders were more successful at colonizing after burning, which provided an initial advantage in resource competition for these invasive alien arthropods (Malumbres-Olarte *et al.*, 2014). Intensified fires along with overgrazing led to the loss of fire-intolerant trees and shrub species and facilitated the establishment of the alien grass *Bromus tectorum* (downy brome) in shrublands and increased establishment and spread of invasive alien species in coniferous forests (Chambers *et al.*, 2007, 2014; Keeley *et al.*, 2003; Roundy *et al.*, 2014).

Some invasive alien species not only benefit from modified fire regimes, but once established, may in turn further modify fire behaviour and community composition (Grace *et al.*, 2001). For example, some alien species increase the fuel load or flammability of the ecosystem, resulting in increased fire frequency or intensity (Brooks *et al.*, 2004; Mandle *et al.*, 2011). Alternatively, burned environments may be more susceptible to biological invasion by species with fire specific traits, such as seed release contingent upon fire or smoke (Franzese & Raffaele, 2017; Gaertner *et al.*, 2014). In many cases, fire primarily influences native and alien animal species through effects on vegetation structure and composition. For example, within Australian tropical savannahs, invasive alien predators such as feral cats, dingoes and snakes may be attracted to burnt landscapes, where they may hunt more effectively (Lozon & MacIsaac, 1997; H. W. McGregor *et al.*, 2014, 2016). In Australian forests, prescribed forest fires reduced understory cover by more than 80 per cent and the occurrence of invasive alien predators increased five-fold, whereas medium-sized native mammalian prey were disadvantaged (Hradsky *et al.*, 2017). Climate and land-use change are now driving changes in global fire regimes, pointing to potentially important interactive effects (**section 3.3.4**).

3.3.1.5.3 Aquatic and marine disturbance regime changes

The biological invasion of aquatic alien species may be facilitated through land-use intensification in the watershed. There is an increased likelihood of invasive alien species establishing within impacted watersheds where increased sedimentation, altered flow rates, increased pollution and habitat destruction lower intrinsic biotic resistance (Havel *et al.*, 2015). In New Zealand braided river systems, alien and native aquatic plants respond to different drivers of change in nature, and in particular disturbances to flow regimes. For example, winter flow variability may increase alien species, while flow stabilization may promote coverage of such species (Brummer *et al.*, 2016). Regulated flow regimes, including floods, also mediate invasive alien plants in the Australian Murray-Darling river catchments (Catford *et al.*, 2011, 2014), with impacts mediated by seasonality along with species' life-history strategies (Greet *et al.*, 2013). In China, the rate of onward spread and establishment of *Lithobates catesbeianus* (American bullfrog) from the points of introduction (for aquaculture) throughout the local watershed was facilitated by native habitat loss and land-use change (X. Wang *et al.*, 2022). In the Mediterranean Sea, fish overgrazing and sediment disturbance caused by vessel anchoring decrease the resistance of native seagrass beds, and thereby allow the establishment and spread of invasive alien *Caulerpa racemosa* (green algae; Tamburello *et al.*, 2014). In Italy, however, clearance of seagrass *Posidonia oceanica* (Neptune grass) led to a reduction in the apparent ecological resistance towards the *Caulerpa cylindracea* (green algae) invasion (Casoli *et al.*, 2021; Marín-Guirao *et al.*, 2015).

3.3.1.6 Landscape and seascape degradation

Anthropogenic degradation of terrestrial ecosystems occurs in nearly all types of landscapes around the world, although there are no consistent global figures about the extent of this phenomenon (IPBES, 2018b). Landscape and seascape degradation involves many processes that drive the decline of biodiversity and ecosystem functioning, and in some cases nature's contributions to people in many parts of the world (IPBES, 2018b; Millennium Ecosystem Assessment, 2005a; United Nations Convention to Combat Desertification, 2017). This section summarizes the roles of desertification, soil/water salinization, soil/water erosion and soil/water acidification in affecting biological invasions.

Land degradation related to desertification is a pervasive global phenomenon in arid and semiarid ecosystems subjected to overgrazing (**section 3.3.1.5**), fire (**section 3.3.4.5**) and drought (**section 3.3.4.2**). Separating out the influence of land degradation *per se* from other interlinked drivers responsible for biological invasions is difficult (IPBES, 2018b; Ravi *et al.*, 2009). Acceleration of soil erosion due to agriculture and mismanagement is widely reported, especially from Asia, Latin America and Africa (FAO, 2015; IPBES, 2018b). Soil erosion can increase the establishment of alien species, as exemplified by annual *Bromus* grass in arid and semiarid grasslands of the United States (Germino *et al.*, 2016).

Additionally, invasive alien species may be intentionally introduced to restore degraded land, for example Indigenous Peoples and local communities report the intentional introduction of invasive alien species to manage land degradation (**section 3.2.5**, **Box 3.6**). Stress-tolerant alien plants are often planted in degraded areas for ecosystem restoration (**Glossary**), and can spread from the planted areas (Hobbs *et al.*, 2006), for instance, alien grasses (e.g., *Bromus tectorum* (downy brome)) seeded for preventing soil erosion in degraded grasslands or for livestock forage in overgrazed rangelands (D'Antonio & Meyerson, 2002; D'Antonio & Vitousek, 1992) and alien trees have been planted to stabilize riparian zones. Irrigated land damaged by salinization is estimated globally to be 60 million ha (FAO, 2015; IPBES, 2018b; Squires & Glenn, 2011). Soil salinization associated with waterlogging leads to the replacement of pre-existing native perennial herbaceous plants with salinity-tolerant alien annual plants in southern Australia (Hobbs *et al.*, 2006).

Especially in North America, Europe and Australia, estuarine and coastal areas have been dramatically transformed over the past 150 to 300 years. Degradation linked to industrialization/urbanization (**sections 3.2.2.4** and **3.3.1.4**) has resulted in accelerated establishment and spread of invasive alien species in once diverse and productive areas (Lotze *et al.*, 2006). For example, creation of a metropolitan coastal front in Athens and the Piraeus port led to the occupancy of an alien scleractinian coral, *Oculina patagonica*, in shallow coastal habitats of the Mediterranean Sea (Salomidi *et al.*, 2013). Overexploitation, pollution, disease and climate change are causing global declines of coastal lagoons and coral reefs, especially on the Great Barrier Reef (IPBES, 2018d). There however seems to be clear evidence that coastal disturbance (e.g., harbour constructions) could be a driver facilitating biological invasions (Boudouresque & Verlaque, 2012; J. Klein *et al.*, 2005).

In aquatic systems, invasive alien species may become more prevalent as construction of water storage infrastructure increases (**section 3.3.1.4**), and in many cases, the mechanisms are linked to changes in disturbance regimes. Features such as reservoirs may act as stepping stones for invasive alien species by providing a homogenous, species-poor, early successional habitat providing little biotic resistance for initial occupation (Havel *et al.*, 2005, 2015; **section 3.4.2**). In the Laurentian Great Lakes region of Wisconsin, United States, Johnson *et al.* (2008) found dams and impoundments to be a significant predictor of occurrence for the invasive alien species *Bythotrephes longimanus* (spiny water flea), *Dreissena polymorpha* (zebra mussel), *Osmerus mordax* (rainbow smelt), *Orconectes rusticus* (rusty crayfish) and *Myriophyllum spicatum* (spiked watermilfoil) – ranging between 2.4 to 300 times more likely to occur in impoundments than natural lakes, even after taking into account other environmental and anthropogenic factors. In the same systems, impoundments were also more likely to support multiple invasive alien species (P. T. Johnson *et al.*, 2008).

3.3.2 Direct exploitation of natural resources

Direct exploitation of natural resources includes both the exploitation of biotic resources through species harvesting (**section 3.3.2.1**) as well as of abiotic resources such as water (**section 3.3.2.2**) and mining for minerals and fossil fuels (**section 3.3.2.3**). These changes are closely linked with major economic (**section 3.2.3**) and demographic (**section 3.2.2**) indirect drivers of change in nature and may lead to a range of wider ecosystem impacts, including habitat degradation and loss as well as changes in landscape and seascape disturbance regimes (**section 3.3.1**). Section 3.3.2 thus describes evidence for links between the extraction of specific resources and invasive alien species **(Figure 3.17)**, and makes reference to other indirect and direct drivers when relevant. The demographic and economic background of these changes are described in **section 3.2** and consequences of land degradation more generally is described in **section 3.3.1**.

3.3.2.1 Species harvesting

Globally, the extraction of biological resources for human use for food, fibre and fuel has doubled since 1970, now constituting more than 22 billion tonnes per year (IPBES, 2019). The livelihood of over 350 million people depends on the extraction of non-timber forest resources, and over six million tons of medium to large wild animals are harvested every year in the tropics, where they are often an important food source (IPBES, 2019). Both terrestrial and freshwater ecosystems are affected by species harvesting in different ways. About 50,000 wild species are harvested for food, energy, medicine, materials, income generation, or other uses globally, and an assessment of 10,098 species

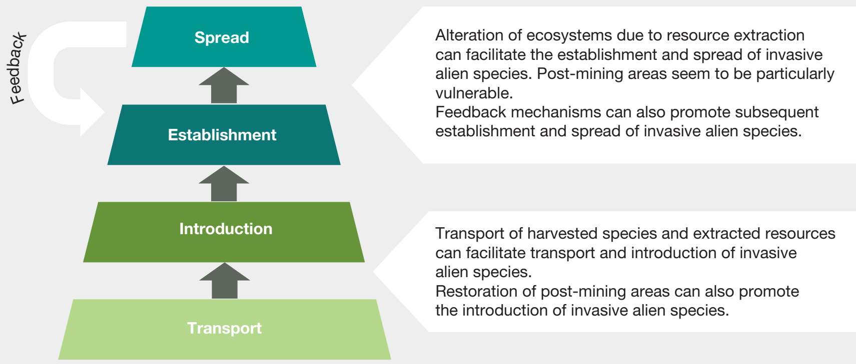


Figure 3 17 **Examples of the role of direct exploitation of natural resources as a driver facilitating invasive alien species across stages of the biological invasion process.**

Transport of harvested species and efforts to restore degraded areas are examples of activities that can lead to the transport and introduction of alien species, the on-site activities related to extraction can facilitate establishment and spread. Introductions of alien species for harvesting often occur at a massive scale, increasing the risk of feedbacks facilitating further introductions.

across 10 taxonomic groups shows that at least 34 per cent are used sustainably, whereas unsustainable harvesting contributes towards elevated extinction risk for 28-29 per cent of near-threatened and threatened species (IPBES, 2022c).

The loss of native biomass and biodiversity in an ecosystem through harvesting has been directly and indirectly linked to increased susceptibility to biological invasions by a wide range of alien species and in a wide range of terrestrial and aquatic environments (IPBES, 2022c; Iverson *et al.*, 2019; Kota *et al.*, 2007; Rebbeck *et al.*, 2017; Schrama & Bardgett, 2016). Many studies of the role of overharvesting on the success of invasive alien species explore indirect effects through trophic changes (Barrios-O'Neill *et al.*, 2016; Kota *et al.*, 2007; Rand & Tscharntke, 2007). For example, an extensive study in the Mediterranean Sea shows that the loss of predators through overfishing resulted in an increase in alien invertebrates in the prey fauna (Rilov *et al.*, 2018). Success of invasive alien species and their impacts may vary, however, depending on whether the harvested native predators are generalists or specialists, and on the intensity and nature of the interaction (Rand & Tscharntke, 2007; W. E. Snyder & Evans, 2006; Tylianakis *et al.*, 2008). Loss of hunted and fished native species may also motivate the release of alien species as alternative species for harvest, as discussed below for terrestrial and aquatic settings.

3.3.2.1.1 Introduction of game for hunting

Stocking of game for hunting purposes is a common practice in many parts of the world, for both recreational and subsistence uses (**section 3.2.1**, **Box 3.2**), and can be an important driver facilitating invasive alien species in many different contexts. A European study (Carpio *et al.*, 2017) found stocking for hunting to be a dominant source of mammal introductions (24 per cent of all known introductions) and birds (30 per cent). Similar patterns were observed in Latin America, where 39 per cent and 22 per cent respectively, of the introduced mammals (69 species in total) and birds (62 species), were introduced intentionally for hunting purposes, compared to an overall 11.2 per cent for food and feed, 5.3 per cent for biological control, and 4.2 per cent for fur industry (Carpio *et al.*, 2020). In the United States, game ranches are also sources of invasive alien species (Geist, 1985). Common alien species introduced for sport or game hunting include: deer species in New Zealand (C. M. King & Barrett, 2005), Australia (N. E. Davis *et al.*, 2016), Latin America (Petrides, 1975) and Europe (Carpio *et al.*, 2017); *Sus scrofa* (feral pig) native to Eurasia and now present on all continents but Antarctica, along with many oceanic islands (Barrios-Garcia & Ballari, 2012; Long, 2003); and various bird species such as pheasants (Blanco-Aguiar *et al.*, 2008) and *Anas platyrhynchos* (mallard) in Australia, New Zealand, South Africa and Hawaii (Rhymer *et al.*, 1994; Fowler *et al.*, 2009; Guay & Tracey, 2009; Government of the Republic of South Africa, 2016). Attempts to control invasive alien species introduced for hunting using biological control have led to further biological invasions, for example the introduction of mustelids to control rabbits and hedgehogs to control garden pests in New Zealand (C. M. King & Forsyth, 2021).

3.3.2.1.2 Introductions of aquatic and marine species for fisheries and angling purposes

Intentional release of fishes and other marine organisms into rivers, lakes and seas to enhance recreational fishing as well as livelihoods is both widespread and common (**section 3.2.1**). These introductions can be distinguished from those linked to aquaculture by being intentionally released into the wild, although the high permeability of aquaculture installations suggests the distinction is somewhat arbitrary (**section 3.3.1.1.1**, FAO, 1995; Grosholz *et al.*, 2015). Globally, hatchery-reared juveniles of more than 180 species of fish and shellfish have been released in the wild for various purposes, including replacing locally extinct stocks (restocking), augmenting a viable fishery (stock enhancement) and creating new fisheries (sea ranching) (Bartley & Bell, 2008; Kitada, 2018; Q. Wang *et al.*, 2006). As many of these releases are occurring at massive scales either outside the native distribution of the species or facilitating spread outside the native range, this constitutes a substantial pathway for aquatic introductions. Introductions, unintentional as well as intentional (e.g., live bait) and legal as well as illegal, have led to the establishment and spread of alien species in freshwater systems in Europe, North America, South America, South Africa and Oceania (Britton & Orsi, 2012; Cambray, 2003; Carpio *et al.*, 2019; Cerri *et al.*, 2018; A. J. S. Davis & Darling, 2017; Ellender & Weyl, 2014; Gherardi *et al.*, 2009; Lintermans, 2004; V. R. Ribeiro *et al.*, 2017; E. R. C. Smith *et al.*, 2020; M. R. Snyder *et al.*, 2020; Weyl *et al.*, 2020). In the Mediterranean basin alone, stocking (for angling, commercial purposes, or biological control) is implicated in over 35 per cent of more than 500 documented freshwater fish introductions (Tricarico, 2012). In the Arctic, *Paralithodes camtschaticus* (red king crab) was intentionally introduced from the Sea of Okhotsk to the Barents Sea in the 1960s to establish a new commercial fishery, and it is currently established and is spreading to the extent that it is commercially harvested by Russian and Norwegian fisheries (Hindar *et al.*, 2020). *Oncorhynchus gorbuscha* (pink salmon) was introduced to several rivers in North-west Russia in the 1950s, and while the first return to rivers in Russia and Northern Norway was recorded in the 1960s, a self-sustaining population did not establish until several decades later. From 2017 onwards, rapid establishment and spread occurred, to the extent that the pink salmon was a dominant fish species in several rivers in Northern Norway in 2021 (with up to 23-fold increase in population size from 2019-2021), and it had spread along the entire Norwegian coast (Berntsen *et al.*, 2022; Hindar *et al.*, 2020).

A compelling example of the cascading effect of species harvesting is the loss of almost 200 species of endemic cichlids following overfishing and introduction of the predatory alien fish *Lates niloticus* (Nile perch) into Lake Nabugabo and Lake Victoria in Africa (Bwanika *et al.*, 2006; Rahel, 2002). Both overharvesting of native species and altered abiotic conditions allowed alien fishes to become established in the lakes, which then eliminated the native species through competition or predation (B. E. Marshall, 2018; Rahel, 2002). The cascade of ecological interactions leading to the demise of native fish in Lake Victoria started with overfishing in the first half of the twentieth century (Aloo *et al.*, 2017), followed by a series of introductions of the invasive alien fish *Lates niloticus*. The *Lates niloticus* population in Lake Victoria peaked at around 2.3 million tonnes in 1999, when it accounted for 92 per cent of the total fish biomass, but fell to less than 300 000 tonnes in 2008, of which the majority were below the required length for export. *Lates niloticus* has subsequently depleted its native prey, hypochromine cichlid fishes (IPBES, 2019), and unsustainable fishing in the lake continued (Luomba, 2016). This top-down cascade led to profound changes in the lake ecosystem, resulting in further reduction in population size and extinction of a number of endemic fishes (**Chapter 4**, **Box 4.10**; B. E. Marshall, 2018).

3.3.2.2 Hydrological resource harvesting

Global water use has increased six-fold over the last 100 years and recent increases in water use have been at a rate of 1 per cent per year (United Nations, 2020). This increasing water use has required large investments in infrastructure, including 50,000 dams and over 16 million reservoirs worldwide (IPBES, 2019), as well as extensive extraction of groundwater resources (International Groundwater Resources Assessment Centre, 2018). There is an increasing recognition that human-mediated hydrological disturbances directly or indirectly facilitate plant and animal invasions (Brummer *et al.*, 2016; Richardson, Holmes, *et al.*, 2007; Truscott *et al.*, 2006). As many alien species thrive in low-competition environments created by hydrological disturbances, biological invasions are often positively associated with the level of hydrological and other disturbances (M. A. Davis *et al.*, 2000; Ricciardi *et al.*, 2017).

Damming and channelization of freshwaters (streams and rivers) and their associated reservoirs can facilitate biological invasions through several mechanisms. First, hydrological alterations through dam constructions may act as reservoirs for invasive alien species and create new habitats which may be colonized by invasive alien species (Richardson, Holmes, *et al.*, 2007). Second, water diversions create new hydrological connections that can facilitate the transfer of a broad suite of aquatic species (including invasive alien species) into new regions. For example, the Chicago Area Waterway was constructed more than 100 years ago to connect Lake Michigan and the Mississippi River, and has permitted invasive alien species to move south from the Great Lakes, and may allow invasive alien *Hypophthalmichthys molitrix* (silver carp) and *Hypophthalmichthys nobilis* (bighead carp) to spread in the opposite direction (e.g., Jerde *et al.*, 2013). Similarly, the water supply to both Los Angeles and San Diego from the lower Colorado River below Lake Mead has been colonized by the biofouling *Dreissena rostriformis* *bugensis* (quagga mussel; e.g., Hickey, 2010). Further, the South-to-North Water Transfer Project, which diverts water from the Yangtze River to northern China, is predicted to promote the further spread of an array of aquatic invasive alien plants, including *Alternanthera philoxeroides* (alligator weed), *Pontederia crassipes* (water hyacinth) and *Pistia stratiotes* (water lettuce) into northern waterbodies (D. Liu *et al.*, 2017).

Hydrological resource use that causes periodic rise and fall of (surface and ground) water levels can make space and resources available for invasive alien species to establish and spread in aquatic and adjacent terrestrial habitats. An example is the invasions of African grasses, including *Melinis minutiflora* (molasses grass), in waterlogged Neotropical savannahs which were driven by changes in the groundwater depth from hydrological disturbance (Xavier *et al.*, 2017). Once an invasive alien species establishes, positive feedback mechanisms, occurring *via* biotic facilitation by the invasive alien species (**section 3.3.5.1**), can promote subsequent biological invasions and promote further spread of invasive alien species, as has been observed for riparian habitats in Czech Republic in which an invasive plant, *Heracleum mantegazzianum* (giant hogweed), resulted in extensive spread in adjacent terrestrial landscapes (Pyšek *et al.*, 2008).

Hydrological alterations or disturbance (e.g., water abstraction) sometimes occur concurrently with other altered habitat conditions (e.g., dryness, salinization, erosion and land and sea degradation) that also favour the introduction and further establishment of invasive alien species. Such alterations in aquatic habitat have favoured establishment of invasive alien species such as *Dreissena polymorpha* (zebra mussel) and *Potamocorbula amurensis* (Amur River clam) in high numbers, with negative consequences for many pelagic and benthic fauna species, especially native mussels in those ecosystems (Grosholz, 2002).

3.3.2.3 Mining (minerals, metal, fossils fuels)

Mining for minerals, metals, oil and other fossil fuels is driven by the energy demands of modern society (Ali *et al.*, 2017) and the transport of these materials accounts for 30 per cent of maritime traffic (IUCN, 2020) contributing to 60 per cent of global GDP (IPBES, 2019). Resource extraction activities also assist in the introduction of invasive alien species to new locations. While mining has a relatively small contribution to overall land-use change (less than about 1 per cent of the area; Maus *et al.*, 2020), its ecological footprint is large (Sonter *et al.*, 2014).

The transport of equipment, the construction of roads or harbours to access mining sites, and the associated increase in vehicles or ships for construction of infrastructure and for transport of mining products can act as pathways and vectors for the introduction of invasive alien species and pathogens, while also facilitating the establishment and spread of invasive alien species due to increased disturbance (F. Bell & Donnelly, 2006; Gelbard & Belnap, 2003; **sections 3.3.1.3**, **3.3.1.4**, **3.3.1.5** and **3.3.1.6**). For example, vehicles and road drainage assisted in the dispersal of spores of *Phytophthora lateralis*, an alien root disease, known to infect the native *Chamaecyparis lawsoniana* (Port Orford cedar) in coal mining areas in the United States (Zobel *et al.*, 1985). Similarly, the introduction of marine invasive alien *Tubastraea* spp. (sun corals) to Brazil was associated with towing and anchoring of oil platforms in coastal waters (Capel *et al.*, 2019). The disturbance and disruption to landscapes caused by mining and resource extraction creates suitable habitats for alien grasses, shrubs and trees to establish and spread (Franklin *et al.*, 2012; Lemke *et al.*, 2013). This can occur across several different types of mining such as coal bed mining (Bergquist *et al.*, 2007; Oliphant *et al.*, 2017), open pit mines (Hou *et al.*, 2019) and fossil fuel extraction activities (Butt *et al.*, 2013). In the context of mining, increased abundance of established invasive alien species are associated with disturbed and fragmented of habitats, the removal of native vegetation cover or through the altered soil nutrients from mine-water discharge (Bergquist *et al.*, 2007).

Mining increasingly tends to occur in remote and previously undisturbed areas (Butt *et al.*, 2013), which may enhance the potential contribution of mining to invasive alien species colonization and establishment in new areas globally. For example, there has been a significant increase in the exploration of oil and gas in marine environments (Jouffray *et al.*, 2020), particularly in Africa (G. Zhang *et al.*, 2019) and the Arctic and Antarctic regions (Petrick *et al.*, 2017), increasing the risk of biological invasions in the marine realm, across multiple taxa, from shipping (Seebens *et al.*, 2013) and discharges of ballast water (Holbech & Pedersen, 2018). Multiple regions and ecosystems across the world are at risk of increased mining activity, including forests (Macdonald *et al.*, 2015), mangroves (Numbere, 2019), the Arctic (Vestergaard *et al.*, 2018) and oceans (Pirotta *et al.*, 2019).

The restoration of post-mining landscapes may also act as a potential pathway for the introduction of alien and invasive alien species. For example, several studies highlight the use of alien species in restoration activities (Mayonde *et al.*, 2015; Oliphant *et al.*, 2017). In South Africa, alien *Tamarix* (tamarisk) species, which have been used in the restoration of post-mining landscapes, have since hybridized with indigenous *Tamarix* species, posing a potential risk for future biological invasions by hybrids (Mayonde *et al.*, 2015; **section 3.3.5.1**). In the United States, the planting of the alien shrub *Elaeagnus umbellata* (autumn olive) has resulted in spread of the species beyond the sites of introduction, which hinders vegetation recovery (Oliphant *et al.*, 2017). Post-mining areas may also be considered as potentially suitable sites for the cultivation of biofuel feedstocks from known invasive alien species, which may spread beyond the area of introduction due to the inherent invasive traits of the species. While the outcomes of introducing known invasive alien species to improve soil conditions where native vegetation is unable to grow due to contamination is beneficial, introducing new species to areas where they were not before (Prabakaran *et al.*, 2019) may increase the risk of alien species spreading to new areas. A meta-analysis of small mammal recovery in passive and actively restored mining areas (Lawer *et al.*, 2019) found that the abundance of invasive alien species was significantly higher in actively restored areas compared to native species. Similarly, studies on the vegetation composition of four coal mines in the Yunnan Province, China, found that invasive alien species occur mainly in degraded areas where active mining or restoration activities are taking place (Hou *et al.*, 2019). However, invasive alien species occurrence can also be influenced by the type of mine and the material extracted (Hou *et al.*, 2019). In a review of the recovery of post-mining landscapes from North America, Europe and Australia, Macdonald *et al.* (2015) found that native species recovery was slowed by the establishment of invasive alien plants.

The evidence for mining as a driver of change in nature that facilitates biological invasions reported in this section stems mainly from terrestrial and marine realms and from Europe and the Americas. Most evidence exists for the introduction, establishment and spread of plants and trees, with little representation of other taxa apart from mammals.

3.3.3 Pollution

Pollution entails releasing new chemical or physical substances or increasing the level or concentration of already-existing substances into ecosystems. Although there are no consistent global assessments on the increase and impacts of pollution, it is believed that pollution has increased at rates similar to the total population growth (IPBES, 2019). Pollution can facilitate invasive alien species through increasing nutrient and resource levels available in ecosystems, such as is the case with eutrophication (**section 3.3.3.1**), through introducing new chemical substances in water or soil (**section 3.3.3.2**) and through dispersal of solids (**section 3.3.3.4**). Marine debris (notably, plastics) are treated separately (**section 3.3.3.3**) because they are a major emerging issue in the context of biological invasions. These pollution sources stem from major economic (**section 3.2.3**) and demographic (**section 3.2.2**) indirect drivers. **Section 3.3.3** thus describes evidence for links between specific pollutants and invasive alien species (**Figure 3.18)**, and makes reference to other indirect and direct drivers when relevant, whereas the demographic and economic background of these changes are described in **section 3.2**.

3.3.3.1 Eutrophication and nutrient deposition

Eutrophication refers to the increase of macronutrients, primarily nitrogen and/or phosphorus in the environment. Major sources are fertilizer use, runoff from animal husbandry and combustion by-products (Stevens, 2019). Common pathways for eutrophication are atmospheric deposition (a major source of oxidized nitrogen) and

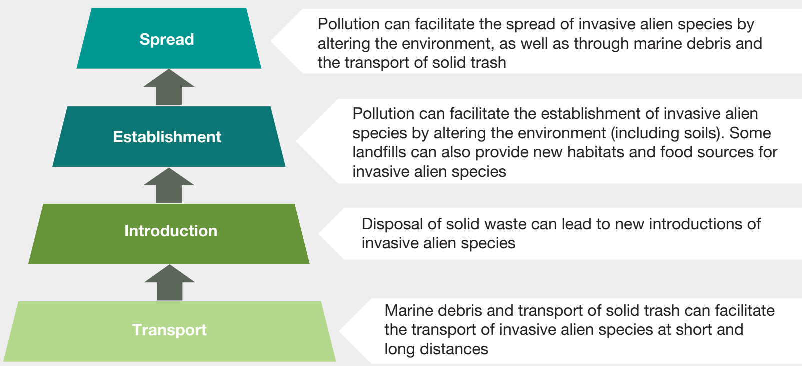


Figure 3 18 **Examples of the role of pollution as a driver of change in nature that facilitates invasive alien species across stages of the biological invasion process.**

Solids (marine debris, solid waste) can facilitate the transport and introduction of invasive alien species and provide habitat and surfaces where alien species can establish and spread. Nutrient and chemical pollution can change habitat quality, making sites more hospitable to invasive alien species.

run-off (nitrogen, phosphorous and other macronutrients, particularly affecting freshwater and coastal systems). The global use of fertilizers increased linearly during the second half of the twentieth century (IPBES, 2019), and for many regions a further increase in the balance of both nitrogen and phosphorus in the soil is expected in the following decades (FAO, 2015). Accordingly, during the last decade alone, there was a four to 20 fold increase in nitrogen flux in aquatic ecosystems (IPBES, 2019). Initial emissions are driven by economic (**section 3.2.3**) and demographic drivers (**section 3.2.2**), as described in the IPBES Global Assessment (IPBES, 2019). Pollution may interact with land- and sea-use change (**section 3.3.1**) and climate change (**section 3.3.4**) in driving the establishment and spread of alien species.

Invasive alien plant species often originate from relatively nitrogen-rich habitats (Dostál *et al.*, 2013), and are hence hypothesized to be more likely to establish and spread under increased nitrogen availability. In line with this, replicated experiments across hundreds of sites in Arctic, boreal, temperate, Mediterranean and tropical grasslands demonstrate that nitrogen addition generally increases the occurrence, abundance, and impact of invasive alien plants, a pattern that is especially pronounced towards warmer and wetter sites (Borer *et al.*, 2017; Seabloom *et al.*, 2013). A review from the United States reported that in temperate and Mediterranean temperate vegetation, increased abundance of invasive alien species is consistently documented as an impact when nitrogen deposition exceeds the critical load for that vegetation type (Pardo *et al.*, 2011). Nitrogen deposition and eutrophication have also been associated with increased abundance of alien tree and understory plant species in a wide range of other habitats, including tropical humid forests (Cusack *et al.*, 2016), temperate forests (Gilliam, 2006) and European lowland heaths (Fagúndez, 2013). Urban and other strongly human-impacted areas may be especially susceptible to invasive alien species facilitated by eutrophication (Ladd, 2016). Eutrophication may also interact with and feedback to other drivers in facilitating invasive alien species, for example, several reviews find that across Mediterranean-type ecosystems, nitrogen deposition favours alien grasses, which accumulate dead biomass in the dry season, increasing wildfire risk (Ochoa-Hueso *et al.*, 2011; Ochoa-Hueso & Manrique, 2010; Vasquez *et al.*, 2008; **section 3.3.1.5.2**).

In aquatic systems, a global study of invasive alien chironomids found that a high proportion of the reported cases were from eutrophic waters, including anthropogenic urban lakes and drainage channels and wastewater treatment plants (Linders *et al.*, 2020). Eutrophication also facilitates alien algae invasions in European lakes and streams (Wilk-Woźniak & Najberek, 2013) and aquatic plant invasions in China (Wu & Ding, 2019). A meta-analysis of drivers of change in freshwater systems revealed surprisingly few studies that explicitly consider alien species (Alahuhta *et al.*, 2019). A review of over 400 marine algal invasions found that most invasive alien species were encountered in eutrophicated waters, but also points out that several vectors and disturbances correlate with eutrophication, and that there is a general lack of experimental studies so it is difficult to establish causality (S. L. Williams & Smith, 2007). A well-studied biological invasion by the potentially toxic dinoflagellate *Prorocentrum minimum* in the Baltic Sea was empirically linked to eutrophication (Hajdu *et al.*, 2005). Eutrophication caused by aquaculture can be particularly conducive to macroalgal invasion in former seagrass beds (Boudouresque *et al.*, 2021; Gennaro & Piazzi, 2011). Extreme eutrophication in aquatic systems can lead to hypoxia. Upon recovery, empty niches may be filled with opportunistic invasive alien species, as shown for an invasive alien nematode which dominates areas of the recovering Ems estuary on the border between the Netherlands and Germany (Essink, 2003).

Studies of eutrophication as a driver affecting invasive alien species are most often focused on plants and algae as they are autotrophic and take macronutrients up directly from the environment. Such studies focus on the establishment and spread phase of the biological invasion process and most focus on alien species of vascular plants in Europe and North America (the geographic bias in these studies largely reflecting overall bias in the location of scientific studies). The available evidence suggests that the role of eutrophication in driving biological invasions is often variable, and both species- and system-specific. On one hand, alien species that respond positively to nitrogen availability in their native range also tend to respond positively to high nutrient availability in their invaded ranges (Borer *et al.*, 2017). On the other hand, in the absence of extrinsic nutrient addition, or in systems where nutrient addition *per se* does not lead to increased productivity or growth, such as for example in tropical forests, alien species can nevertheless benefit from nutrient addition (Cusack *et al.*, 2016). Marine studies are particularly focused on the spread of invasive alien species in Mediterranean seagrass ecosystems.[7] Authors found no studies directly linking eutrophication to terrestrial vertebrate invasions, despite the potential for a link between soil nitrogen and invasive alien herbivores *via* bottom-up processes.

3.3.3.2 Other contaminants in water and soil

Human modification of environments due to high pressure on direct exploitation of natural resources (**section 3.3.2**), land-use change (**section 3.3.1**) and urbanization (**section**

7. Data management report available at https://doi.org/10.5281/zenodo.5529309

3.2.2.4) has led to deposition of diverse contaminants in soil and water. Construction and maintenance of roads introduces metals (especially lead, but also aluminium, iron, cadmium, copper, manganese, titanium, nickel, zinc and boron), salts, ozone and nutrients into roadside environments (Trombulak & Frissell, 2000). This introduction creates an opportunity for alien species that are highly tolerant to contaminants to establish in areas where native species are struggling. For example the roadside *Melinis repens* (natal redtop) is a common naturalized species in Australia and high levels of trace metal were found in its tissues (C. Pratt & Lottermoser, 2007). Zhang *et al.* (2008) found that higher tolerance to lead stress enabled the alien *Sambucus canadensis* (American black elderberry) to outperform the native *Kummerowia striata* (Japanese lespedeza) and may have promoted its rapid establishment in lead contaminated soil. In Brazil, in a rocky neotropical savannah, Barbosa *et al.* (2010) found that paved roads, by reducing aluminium toxicity, favour alien species and helps them during the first stages of the biological invasion process. However, in some cases, soil contamination can also limit the fitness of invasive alien species and influence the dynamics between invasive alien and native species. De la Riva and Trumble (2016) investigated the effect of selenium on reproduction and competitive behaviour of the invasive alien *Linepithema humile* (Argentine ant) and found environmental toxins may not only pose problems for native ant species, but may also serve as a potential obstacle for establishment among alien species; *Linepithema humile* reproduced less when exposed to selenium.

Pollutants in aquatic systems, including metals contained within antifouling paints, can enhance the establishment success of invasive alien species (Piola & Johnston, 2008), particularly those that have a positive association with metal contamination such as the invasive alien hull-fouling bryozoan *Watersipora subtorquata* (McKenzie *et al.*, 2012). Additionally translocations of static maritime structures and movement of semi-submersible rigs continue apace in the Anthropocene and so act as largely overlooked and unregulated vectors of marine invasive alien species (Iacarella *et al.*, 2019; Wanless *et al.*, 2010). Also, water pollution caused by high alkalinity and nitrate concentration (**section 3.3.3.1**) is associated with the occurrence of aquatic invasive alien species that are among Europe's top 10 invasive alien species, such as *Dreissena polymorpha* (zebra mussel), *Procambarus clarkii* (red swamp crayfish) and *Salvelinus fontinalis* (brook trout) (Gallardo, 2014).

3.3.3.3 Marine debris

Marine debris is defined as "any persistent manufactured or processed solid material discarded, disposed of or abandoned in the marine and coastal environment" (Agamuthu *et al.*, 2019), thus dispersal of any alien species through marine debris is considered anthropogenic. The six main categories of marine debris are plastic, paper, metal, textile, glass and rubber (Agamuthu *et al.*, 2019). Plastic comprises 50 to 90 per cent of the total marine debris found globally (Eriksen *et al.*, 2014). A 2014 estimate of the amount of plastic pollution floating on the ocean revealed that there is a minimum of 5.25 trillion particles weighing 268,940 tons (Eriksen *et al.*, 2014). In the absence of further regulations, the amount of plastic entering aquatic ecosystems annually is expected to increase from 14 million tons per year in 2016 to 23-37 million tons per year by 2040 (United Nations Environment Programme, 2021). This burgeoning amount of plastic debris in the ocean has created unprecedented opportunities for the dispersal of marine organisms through rafting, representing a potential mechanism for biological invasions. Floating marine debris can disperse attached organisms significant distances depending on the ocean current speed and direction, and thus facilitates first introductions (*via* long-distance transport) to a new region, and secondary spread (short-distance transport) within an invaded region (Rech *et al.*, 2016). Floating plastic degrades much more slowly than natural rafting material and therefore is a potentially more potent vector for long-distance dispersal of invasive alien species (Agamuthu *et al.*, 2019).

Flotsam and jetsam (floating debris) usually start their floating journey in a "clean" state (i.e., free of fouling biota). Debris provides a new habitat for marine species adding new surfaces for colonization by organisms. Because debris usually spends a long time periods in the marine environment, debris often hosts an extensive and reproductively active fouling biota, before becoming part of marine floating litter (Kiessling *et al.*, 2015). For example, in Colombia, 86 per cent of marine debris is composed by wooden materials and plastic litter, which generate the optimal conditions for species to float away and colonize novel areas. Indeed, this study found that 62 per cent of the surveyed beaches were found to have marine fauna using floating plastic or wood as a substrate for potential rafting and dispersal (Gracia C. *et al.*, 2018). Organisms ranging from algae to reptiles (i.e., iguanas) have been observed to raft on floating objects, but the most common species include barnacles, polychaete worms, bryozoans, hydroids and molluscs. There is evidence for the transport of 270 species belonging to 85 taxa, including at least five invasive alien species on floating objects on the sea; however this phenomenon is likely still underestimated due to the limited number of studies and observations at the species level (Avio *et al.*, 2017). The highest numbers of rafting taxa on floating litter were found in the Pacific and North Atlantic, which might be explained by the overall high research effort undertaken in these regions (Kiessling *et al.*, 2015).

Marine plastic debris is largely attributed to fisheries and leisure or household gear (Gracia C. *et al.*, 2018). Off the

Asturian coast, Spain, rafting biota identified included species of goose barnacles, acorn barnacles, bivalves, gastropods, polychaetes and bryozoan, and hydrozoan colonies attached to stranded litter, many of which were alien species, such as *Magallana gigas* (Pacific oyster) and *Austrominius modestus* (Australian barnacle) (Rech *et al.*, 2018). Plastics, except for foam, sustained a more diverse attached community than non-plastic materials (Rech *et al.*, 2018). Another study carried out in New Zealand identified that the most common biofouling taxa traveling in marine debris were hydroids, bryozoans, algae and polychaetes (Campbell *et al.*, 2017). Off the Cantabrian Coast, alien species expansions could be reinforced by the presence of manufactured objects in the sea. *Austrominius modestus, Magallana gigas*, the potentially invasive alien *Amphibalanus Amphitrite* (striped barnacle) and other 14 species were found attached to plastic bottles and fishing gear, in particular on ropes (Miralles *et al.*, 2018). Off the west coast of Svalbard, a study that assessed the density of macro-plastic litter and the biota established on them found that the largest objects (fishing boxes, containers) were colonized by *Semibalanus* sp. (barnacles), *Lepas* sp. (goose barnacles), *Mytilus* sp. (blue shells), bryozoans and marine macro-algae.

Box 3 8 **The spread of invasive alien species on Japanese tsunami marine debris.**

On the 11 March 2011, an undersea megathrust earthquake struck Japan and created a tsunami on its East coast (specifically in the Tohoku coast of Northeast Honshu, Japan) that reached 38.38 m in height (Carlton *et al.*, 2017; Shimada, 2016). The tsunami produced abundant marine debris and caused the translocations of multiple taxa that were concentrated in the Pacific Northwest of the United States. At least 289 living Japanese coastal marine species have been found since 2012 on the coastlines of North America and Hawaii; the biota included macroinvertebrates, fish, microinvertebrates and protists (Carlton *et al.*, 2017). According to Miller *et al.* (2018), one of the most common species arriving on Japanese tsunami marine debris is *Mytilus galloprovincialis* (Mediterranean mussel). During the following years after the tsunami, various reports associated the appearance of new invasive alien species, not previously reported, with this event. The establishment of rafting species will depend on the number and frequency of reproductively viable individuals being transported on the marine debris coupled with the presence of suitable environments in the recipient range (Carlton *et al.*, 2017). The tsunami occurred early in the breeding season for many coastal species, which may have contributed to a successful settlement on Japanese tsunami marine debris (J. A. Miller *et al.*, 2018).

Figure 3 19 **Marine debris caused by the 2011 tsunami in Japan.**

The derelict was discovered off the coast of Seal Rock, Oregon, USA in April 2015 after having been missing from Japan since the tsunami on 11 March 2011. Photo credit: John W. Chapman – under license CC BY 4.0.

The rafting of groups of adult organisms favours their better biological dispersal compared to larval transport, and is regarded as the main reason for reappearance of the genus Mytilus on Svalbard (Weslawski & Kotwicki, 2018). In the western Mediterranean Sea, plastics were the major type of debris found because of its poor degradability; however, glass, cans, fishing nets and polyurethane containers were also found. Macro-benthos living on raft material comprised mainly molluscs, polychaetes and bryozoans, large fish were found commonly below large plastic bags, while following resources linked with the bags, these fish might move outside of their native range (Aliani & Molcard, 2003). Non-plastic objects, while less abundant and less ephemeral, can still help in spreading invasive alien species, as shown by an example of 10 alien mollusc species found on a single buoy (Ivkić *et al.*, 2019).

Marine debris can interact with other drivers in facilitating biological invasions. Notably, natural disasters can enhance the movement of invasive alien species traveling on marine debris (**section 3.4.1**). For several years following the Japanese tsunami in 2011, debris with living species from Japan has landed on coastlines from Midway Atoll to Hawaii Island and from south central Alaska to central California **(Box 3.8)**. Using the data from this event to model potential establishment, Simkanin *et al.*(2019) found that of 48 invertebrate and algal species on the Japanese tsunami marine debris, 27 per cent (13 species) had landed on Asturian coast locations with suitable environmental conditions for establishment and survival, and a further 43 per cent (21 species) had environmental requirements met in other areas where tsunami debris likely landed (but had not been documented).

3.3.3.4 Dispersal of solid waste

In 2016, humans generated over 2 billion tons of municipal solid waste, and by 2050 this number is predicted to increase to 3.4 billion (Kaza *et al.*, 2018). Solid waste can both transport and sustain a high variety of alien organisms, thus contributing to the spread of invasive alien species. A global review reported the establishment of 215 alien plant species in waste disposal sites (Plaza *et al.*, 2018). In Pakistan, industrial waste increased the recruitment of the invasive alien tropical trees *Prosopis juliflora* (mesquite) and *Leucaena leucocephala* (leucaena) (Uzair *et al.*, 2009). In central Brazil, the dispersal of the invasive alien grass *Arundo donax* (giant reed) seems to be assisted by the disposal of construction waste (Simões *et al.*, 2014). Disposal of garden waste may facilitate the spread of ornamental alien plant species, as examples, in Spain the cactus *Opuntia engelmannii* subsp. *lindheimeri* (Lindheimer pricklypear; Elorza *et al.*, 2004), in Argentina, the fast-growing liana *Podranea ricasoliana* (pink trumpet vine; Hurrell *et al.*, 2012) and the rhizomatous fern *Pteris parkeri* (Cretan brake; Guerrero, 2017). Dumping garden waste in close proximity to watercourses contributed to the spread of *Reynoutria sachalinensis* (giant knotweed) and *Reynoutria japonica* (Japanese knotweed) in riparian habitats in the Czech Republic (Pyšek & Prach, 1996). Irresponsible disposal of fragments of alien aquarium macrophytes and macroalgae may promote their introduction and spread in nature (Cohen *et al.*, 2007; Odom *et al.*, 2014; Vranken *et al.*, 2018). Waste disposal sites are a source of propagules of alien plants that can spread into natural habitats. In central Brazil, savannah adjacent to landfills has ten times more alien species than nearby savannah not adjacent to landfills (Santana & Encinas, 2008). Urban mixed deciduous forest sites in Switzerland close to illegal garden waste dumping areas exhibit over 30 times more alien species than nearby control areas (Rusterholz *et al.*, 2012). In addition, landfill areas used for compost production may contain many alien plants (Vaverková *et al.*, 2020), so that the distribution and use of this compost, for example in agriculture, could promote biological invasions (Pietsch, 2005).

Waste disposal sites are often used as a food source by alien vertebrates found close to urban areas (**section 3.2.2.4**), such as *Rattus norvegicus* (brown rat), *Felis catus* (cat) and *Sus scrofa* (feral pig) (Plaza & Lambertucci, 2017). Food waste has been found to be an important item in the diet of feral cats in Mexico (Ortiz-Alcaraz *et al.*, 2017) and Australia (Hutchings, 2003), and a rubbish tip in Australia supported a high density of feral cats (Denny *et al.*, 2002). Among alien birds, food waste consumption has contributed to the establishment and spread of *Threskiornis aethiopicus* (sacred ibis) in the United States (Calle & Gawlik, 2011) and Western Europe (Clergeau & Yésou, 2006), as well as for *Passer domesticus* (house sparrow) in urban sites in Kenya (Imboma, 2014). In eastern Madagascar, the abundance of the invasive alien *Duttaphrynus melanostictus* (Asian common toad) is positively related to the presence of rubbish dumps (Licata *et al.*, 2019). Disposal of green waste containing small alien vertebrates may also contribute to their spread, as was possibly the case during the rapid expansion of *Leiocephalus carinatus armouri* (northern curly-tailed lizard) in Florida (H. T. Smith & Engeman, 2003). In South Central United States, landfills facilitate the establishment and spread of *Paratrechina fulva* (tawny crazy ant; ISAC, 2016). In South America, waste disposal sites provide food and hiding sites for *Lissachatina fulica* (giant African land snail; Gregoric *et al.*, 2013; Kaique & Nara, 2017; Thiengo *et al.*, 2007). Alien species account for 30 per cent of the richness and abundance of macro-snails in landfills in the United Kingdom (Rahman *et al.*, 2016). The accumulation of water in solid waste disposed in urban areas favours the proliferation of alien mosquitos of the genus *Aedes*, which are vectors of several diseases that affect humans (e.g., *Aedes aegypti* (yellow fever mosquito); Baldacchino *et al.*, 2015).

In summary, the disposal of solid waste has contributed to the introduction, establishment and spread of a wide variety of alien plant and animal species in terrestrial and aquatic habitats across continents. Most of the studies took place in Europe and North America. There is a lack of studies in the East Asia and Pacific regions, where the highest amount of solid wastes are produced (Kaza *et al.*, 2018).

3.3.4 Climate change

Anthropogenic climate change has emerged as a dominant threat to Earth's biodiversity and ecosystems over the last few decades, altering species' ranges and abundances, reshuffling biological communities, restructuring food webs, and altering ecosystem functions (IPBES, 2019; IPCC, 2022). Alterations in global temperature and precipitation regimes are predicted to facilitate biological invasions by increasing the likelihood of introduction and establishment of invasive alien species in many areas, thus increasing the potential invaded range of invasive alien species (Hellmann *et al.*, 2008; IPBES, 2019; IPCC, 2022; Walther *et al.*, 2009). Climate change may further facilitate biological invasions by increasing rates of reproduction and survival in sites where invasive alien species are already present, hence facilitating their establishment and further spread (Chown *et al.*, 2012; Fløjgaard *et al.*, 2009; Loomans *et al.*, 2013). The range or population growth rates of some alien species may currently be limited by climatic variables which may become more favourable in the future (e.g., low temperature, low precipitation in regions where these climate factors show increasing trends; Bradley *et al.*, 2010; Ibáñez *et al.*, 2009; O'Donnell *et al.*, 2012). Some studies based on bioclimatic models predict that more frequent extreme events may have potential to trigger or alter the trajectory of biological invasions (Hulme, 2017; Pyšek *et al.*, 2020). While models and projections point to potentially strong impacts of climate change on biological invasions (**section 3.6.3**, **Box 3.14**), empirical data that unambiguously attribute shifts in alien species' distributions and abundances to climate change are rare.

Climate change entails shifts in both mean conditions and the frequency and magnitude of climatic extremes, all of which can have consequences for biological invasions. This section synthesizes knowledge about how invasive alien species are affected by changes in temperatures (**section 3.3.4.1**), precipitation regimes (**section 3.3.4.2**), extreme events (**section 3.3.4.3**), CO_2 concentrations (**section 3.3.4.4**), fire frequencies and magnitudes (**section 3.3.4.5**), sea level rise (**section 3.3.4.6**), and, assisted colonization is an example of a climate mitigation strategy with high relevance for invasive alien species **(Box 3.9)**. Climate change results from major economic (**section 3.2.3**) and demographic (**section 3.2.2**) indirect drivers over long timescales. **Section 3.3.4** describes evidence for links between specific climatic changes and invasive alien species **(Figure 3.20)**, and makes reference to other indirect and direct drivers when relevant. The demographic and economic background or these changes are described in **section 3.2**.

3.3.4.1 Temperature change

Global mean surface temperature is projected to rise between 1.4°C and 4.4°C by the end of the twenty-first

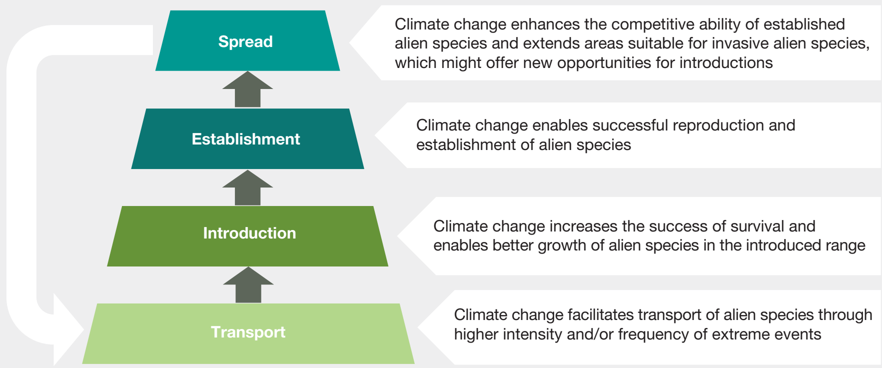


Figure 3 20 **Examples of the role of climate change as a driver of change facilitating invasive alien species across stages of the biological invasion process.**

Extreme events may facilitate transport, changes in climatic conditions may enhance survival, reproduction, and competitive ability, which in turn increases introduction establishment and spread, of alien species. Climate change may extend the geographic ranges potentially suitable for invasive species, providing potential feedback from spread (within the invaded range) to transport (to new areas beyond the current invaded range). Adapted from Walther *et al.* (2009).

Box 3 9 **Assisted colonization.**

Species translocations for conservation purposes have been conducted for decades, but the vast majority are reintroductions or population enhancements of species within their historical range, whereas assisted colonization (variously termed “assisted colonization”, “assisted migration” and “managed relocation”; Hoegh-Guldberg *et al.*, 2008; J. M. Mueller & Hellmann, 2008; Richardson *et al.*, 2009) is concerned with moving species into areas beyond the range in which they have a recent evolutionary history. Assisted colonization could become a significant driver facilitating biological invasions in the future as climate change adaptation and ecosystem restoration strategies increasingly argue for species translocation (D. M. Hansen, 2015; Lunt *et al.*, 2013). Thus far, assisted colonizations of animals have typically involved relatively short-distance translocations, most often to islands (e.g., Freifeld *et al.*, 2016; Griffiths *et al.*, 2010; Scoleri *et al.*, 2020). Since the mid-2000s there have been an increasing number of proposals to intentionally introduce plants and animals to favourable habitats beyond their historical ranges, with the goal of protecting such species against climate change and other environmental stressors (Seddon, 2010).

It has been proposed that decisions regarding assisted colonization schemes can be guided safely by an assessment of the costs and benefits of translocation (Hoegh-Guldberg *et al.*, 2008), including the potential ecological or economic impacts of biological invasions. Others have argued that the ability of ecologists to forecast ecological costs is weak (Ricciardi & Simberloff, 2009). Some researchers suggest that the risks of ecological disruption can be reduced by moving species within the same continent (J. M. Mueller & Hellmann, 2008), especially where closely related species exist. In such situations the translocated species is less likely to encounter communities that lack eco-evolutionary experience with functionally similar taxa and thus the species’ abundance and impact are more likely to be constrained through species interactions; however, this rationale ignores risks of hybridization, competitive displacement and disease transmission (Arcella *et al.*, 2014; Morales *et al.*, 2013; Simler *et al.*, 2019). Given the global influence of climate change as a stressor, the issue of assisted colonization is relevant to biotas in all regions and terrestrial, freshwater and coastal marine realms.

century (2081–2100) relative to 1986–2005, depending on how greenhouse gas emissions develop (Arias *et al.*, 2021; IPCC, 2014, 2021). Warming seems to be strongest at high northern latitudes and with variable rates in mountainous biomes in comparison to lowlands (Loarie *et al.*, 2009; Mountain Research Initiative EDW Working Group, 2015; C. Nolan *et al.*, 2018; Q. Wang *et al.*, 2016). The warming is associated with other changes in ecosystems such as contraction of snow cover and permafrost areas (Luláková *et al.*, 2019) and increased risk of heat and precipitation extremes (IPCC, 2007, 2021) affecting the productivity and water-use efficiency and spatial shifts of habitats (Svenning & Sandel, 2013).

The general expectation is that with increasing temperatures, some established alien species will be able to expand their ranges polewards and to high elevations and thus expand their introduced ranges without additional human assistance. Warming is a major component in forecasts of the responses of 100 of the world’s worst invasive alien species to climate change, according to the International Union for Conservation of Nature (IUCN), who project significant range shifts, and hence further spread of invasive alien species within and beyond their current invaded ranges (Bellard *et al.*, 2013). For example, outbreaks of bluetongue virus, a disease of ruminants transmitted by *Culicoides* species (biting midges), occurred for the first time in northern Europe in 2006 as a result of warmer temperatures. In the future, these northern regions will become increasingly suitable for this midge vector, which could spread unaided on prevailing winds (A. E. Jones *et al.*, 2019). Increased temperatures may benefit the establishment of some alien species regularly intercepted at the border by quarantine officers as contaminants of goods (particularly agricultural and horticultural produce) or stowaways on transport vectors (such as in or on boats). Since the 1970s, the establishment in the United Kingdom of alien invertebrate plant pests intercepted at ports of entry or for which outbreaks have been reported is positively correlated with average winter temperatures, but no such relationship was found for plant pathogens (Hulme, 2017). However, increases in winter temperature was found to facilitate the spread of plant pathogens in North America (Kliejunas, 2011).

Climate change may increase the probability of establishment and spread of alien species that are currently present in a particular region in anthropogenic environments such as buildings, glasshouses, and gardens but are limited by climate from surviving in nature. For example, in the United Kingdom warmer winter temperatures are expected to increase the probability that *Liriomyza huidobrensis* (serpentine leafminer) and the soil-borne *Athelia rolfsii* (sclerotium rot), currently found in glasshouses, will be able to overwinter outside and consequently establish (Baker *et al.*, 1996; Hardwick *et al.*, 1996). Similarly, casual annual C4 weeds (e.g., *Setaria viridis* (green foxtail), *Digitaria sanguinalis* (large crabgrass)) that do not tolerate frost and thus do not currently survive the winter in the United Kingdom may become problematic in arable agriculture in a warmer future when this constraint is lifted, especially as they are well-adapted to high temperatures that some British native plants may not tolerate as well (Froud-Williams, 1996). For insects in temperate regions, a major effect of climate warming is enhanced individual growth and development and consequent increased winter survival, allowing range expansion to more northerly latitudes (Bale *et al.*, 2002). Thus several invasive alien insect species, many of which are crop pests and diseases, may expand their ranges northward and upward under climate change (Lehmann *et al.*, 2020). In aquatic systems, substantial non-breeding populations of the *Trachemys scripta* (pond slider) have persisted for some time in regions where climate change could soon facilitate reproduction and subsequent establishment and spread (Rödder *et al.*, 2009). Similarly, warming of North American lakes is likely to increase thermal suitability for species of fishes currently with a more southerly distribution, including many alien species, that could potentially expand their distribution poleward into alien regions, potentially as far as the Arctic (Ricciardi *et al.*, 2020).

Several empirical studies support and confirm some of these projections. Higher temperatures, extended summer seasons, and increasing available thermal budget for growth are recognized as potential explanations for ongoing polewards shifts in species' distributions (S. C. Mason *et al.*, 2015; Parmesan & Yohe, 2003). There is evidence that alien species may be especially well-suited to exploit opportunities for range expansions offered by warming. For example, over the past two decades, alien plant species in the European Alps have colonized higher altitudes approximately twice as rapidly as native species (Dainese *et al.*, 2017). As another example, the majority of alien species in the Mediterranean originate from the Red Sea (i.e., Lessepsian migrants; about 67 per cent of all alien species; **section 3.3.1.3**, **Box 3.7**), with a small proportion (about 7 per cent) from other tropical areas. These alien species have long been confined to the easternmost Levantine shores, and the warming of the Mediterranean is now facilitating their further spread (Lejeusne *et al.*, 2010). Temperature has been found to limit key performance parameters in alien species across taxonomic groups and regions, including fecundity in mammals (D. J. Bell & Webb, 1991), fish (Fobert *et al.*, 2011) and birds (Shwartz *et al.*, 2009); growth in marine algae (Hales & Fletcher, 1989) and fish (Kornis *et al.*, 2012); survival in amphipods (Ashton *et al.*, 2007; Cowling *et al.*, 2003) and mosquitoes (Roiz *et al.*, 2011) and growth, survival and fecundity in plants (Willis & Hulme, 2002). However, temperature sensitivity in such performance parameters does not automatically translate into increased performance in a warming climate. For example, in the United Kingdom, alien plants have a faster phenological response to warming than co-occurring native species, yet this has not translated into a faster spread (Hulme, 2011b).

As climate change progresses, some regions, biomes and taxonomic groups will be subjected to climates not previously encountered, and invasive alien species are projected to either decrease or increase (Bellard *et al.*, 2013b; **Chapter 2**, **section 2.6**). Bellard *et al.* (2013) project future hotspots of invasive alien species to be in biomes at higher latitudes where future climate change is projected to be less extreme (e.g., temperate mixed forests and woodlands) whereas biomes expected to shift into extreme climatic zones (e.g., tropical forest) may experience a decrease in number of invasive alien species. The ranges of invasive alien terrestrial and aquatic invertebrates, aquatic plants and microorganisms are projected to increase, whereas the ranges of invasive alien amphibians, birds and fungi could experience range contractions under future climate projections (Bellard *et al.*, 2013b; **Chapter 2**, **section 2.6**).

3.3.4.2 Precipitation change

Climate change has caused an increase in global average precipitation since the mid-twentieth century, which has been accelerating since the 1980s, but with great regional and temporal variability, so that precipitation is increasing in some regions, decreasing in others, with interannual variability and seasonality also changing (IPCC, 2021). Few studies explicitly link biological invasions to precipitation change, and effects of precipitation change may relate both to water *per se* and to consequences for disturbance regimes or dispersal. Precipitation extremes cause disturbances which can create open sites suitable for colonization, especially in and along streams where precipitation extremes may also be associated with increased propagule pressure, leading to increased risk of biological invasions (Pyšek, Bacher, *et al.*, 2010). For example, drought and changes in flow regimes of rivers and streams (which are of relevance for precipitation change) can facilitate the spread of invasive alien plant species along streams in Europe, both directly and through negatively affecting the native plant community (Catford *et al.*, 2011, 2014). As another example, invasive alien European *Bromus* spp. grasses in North America can exploit available soil moisture more efficiently and thus recover more rapidly than native vegetation after drought enabling them to invade areas formerly dominated by native woody species following periods of drought (Ricciardi *et al.*, 2020). Precipitation changes may interact with temperature changes in affecting future ranges of invasive alien species. For example, the potential invaded range of *Bactrocera dorsalis* (Oriental fruit fly), a major pest throughout South East Asia that has invaded (and been eradicated from) a number of Pacific Islands and mainland areas in North America and elsewhere is projected to extend further polewards as cold stress boundaries recede, but also contract in areas where precipitation decreases substantially (Stephens *et al.*, 2007). In North America, concurrent changes in precipitation and temperature are projected to extend the potential invaded range for many invasive alien species of forest ecosystems (Dukes *et al.*, 2009). In particular, forest fungal pathogens in

Europe (123 taxa, of which 42 per cent are considered to be alien species) and North America (18 taxa) are sensitive to both low temperatures and drought, and are generally expected to extend their invaded ranges with increasing temperature and precipitation (Dukes *et al.*, 2009; Santini *et al.*, 2013). Precipitation change may also affect the distribution of invasive alien insects. *Solenopsis invicta* (red imported fire ant) was introduced to the United States from sub-Amazonian South America (native range) in the 1930s or 1940s, and has since expanded throughout North America and to Australia and New Zealand along with a number of tropical islands. The potential invaded range of the red imported fire ant is limited by both low temperature and low precipitation, and future projections entail both expansions and contractions, the latter largely in areas where precipitation is projected to decrease (Morrison *et al.*, 2004).

3.3.4.3 Climate extremes

Anthropogenic climate change is causing increasing frequency and/or intensity of climate extremes, including temperature extremes, heavy precipitation and pluvial floods, river floods, droughts, storms (including tropical cyclones), as well as compound events (IPCC, 2021). High-temperature extremes (including heatwaves) have become more frequent and more intense across most land regions since the 1950s, and marine heatwaves have approximately doubled in frequency since the 1980s (IPCC, 2021). The frequency and intensity of heavy precipitation events have increased since the 1950s over most land area, and agricultural and ecological droughts have increased in some regions due to increased land evapotranspiration (IPCC, 2021). Climate extremes can cause dramatic ecosystem destabilization and an abrupt shift towards an alternative ecosystem state (Jentsch *et al.*, 2007), which may affect all stages of the biological invasion process (i.e., transport, introduction, establishment and spread; **Chapter 1**, **section 1.4**) (Hulme, 2017). However, global and quantitative assessments on the response of invasive alien species to extreme climatic events seem to be limited even for generally better-studied taxonomic groups such as plants (Orsenigo *et al.*, 2014) and insects (Bale *et al.*, 2002).

There is evidence that climate extremes (e.g., heavy winds, hurricanes, storms and floods) enhance long-distance transport and spread of invasive alien plants, vertebrates, invertebrates and invasive alien species that are agricultural pests and pathogens (Aylor, 2003; J. K. M. Brown & Hovmøller, 2002; Diez *et al.*, 2012; Hellmann *et al.*, 2008; Nagarajan & Singh, 1990). In terrestrial ecosystems, extreme hurricanes in Northern and Central America resulted in the long-distance spread of invasive alien weeds (Masters & Norgrove, 2010), alien vertebrates (e.g., *Iguana iguana* (iguana) and *Osteopilus septentrionalis* (Cuban treefrog); van den Burg *et al.*, 2019) and diseases (*Xanthomonas axonopodis* (gummosis: grasses); Masters & Norgrove, 2010). The frequency of hurricanes positively relates with the large-scale pattern of spread of alien *Phragmites australis* (common reed) in the United States (Bhattarai & Cronin, 2014).

In freshwater and marine biomes, extreme hydrological events (sometimes caused by strong winds) such as storms and floods may facilitate the transport, spread and establishment of invasive alien aquatic organisms (Anufriieva & Shadrin, 2018). Severe floods may allow fishes to escape from farm ponds and culture cages into natural water bodies (Canonico *et al.*, 2005). The Foe Indigenous People around Lake Kutubu in the Southern Highlands Province of Papua New Guinea have also observed the role of climate extremes in facilitating biological invasions. The shift from artisanal small-scale fishing to fish farming introduced alien fish (e.g., *Cyprinus carpio* (common carp)) and plants (*Pontederia crassipes* (water hyacinth)) to fish farms, which then escaped into Lake Kutubu during heavy rains of 2010-2012 (P. T. Smith *et al.*, 2016).

In polar ecosystems, genomic analyses revealed that *Durvillaea antarctica* (cochayuyo) has recently travelled more than 20,000 km by storm-forced surface waves (or oceanic storm waves) and reached Antarctica from mid-latitude source populations (C. I. Fraser *et al.*, 2018). In subarctic regions, extreme heatwaves cause hypoxia and high water temperatures, which seem to lead to widespread mortality of native freshwater fishes and facilitate the invasion of cool and warm water alien species (Rolls *et al.*, 2017).

Increased incidence and severity of heavy winds may also facilitate increased seasonal northward spread of plant pests and pathogens (Aylor, 2003; J. K. M. Brown & Hovmøller, 2002; Hopkinson, 1999; Olfert *et al.*, 2016, 2017), resulting in biological invasions beyond the current northern range limit of these species. For example, cereal rusts (Pucciniales) typically overwinter on cereals and grasses in the southern United States and northern Mexico, and the spores are blown northward in the spring or early summer by wind currents, affecting winter and spring cereal crops (Eversmeyer & Kramer, 2000; Xi *et al.*, 2015). Increased incidence and severity of heavy winds under climate change may enhance transport of rust species into the United States and Canada, thus facilitating biological invasions (Eversmeyer & Kramer, 2000; Xi *et al.*, 2015).

Extreme climatic events that cause catastrophic and widespread damage to ecosystems often increase the availability of resources such as water, nutrients, space and prey for alien species (Diez *et al.*, 2012; Hellmann *et al.*, 2008). In Australian rainforests, severe cyclones cause catastrophic disturbance by opening canopy gaps, thereby facilitating the rapid recruitment and spread of alien woody-vines (e.g., *Thunbergia* spp., *Mikania micrantha* (bitter vine) and *Turbina corymbosa* (Christmas vine); Camarero, 2019). Hurricane Sandy in the United States caused catastrophic coastal dune erosion and thereafter an alien *Carex kobomugi* (Asian sand sedge) established (Charbonneau *et al.*, 2017). Also, heavy drought can cause fire activity (e.g., 2019-2020 mega-fires in Australia and 2019 California wildfires, **Chapter 1**, **Box 1.4**), resulting in enhancement of the spread of invasive alien trees from plantation forests (**section 3.3.4.5**).

Extreme climatic events may also often stress and cause catastrophic mortality of resident native species, resulting in decreasing biotic resistance of native communities to the establishment and subsequent spread of invasive alien species (Diez *et al.*, 2012; Hellmann *et al.*, 2008; **section 3.3.5**). In semi-arid shrublands of Chile, a study based on 130 years of precipitation data showed that extreme drought, associated with El Niño effects, led to increased alien plant cover at the expense of native plants (Jiménez *et al.*, 2011). Similarly, using a mesocosm experiment across a precipitation and continental gradient between Belgium and Israel, Jentsch *et al.* (2007) found that drier ecosystems showed decreased biomass production after extreme droughts, facilitating invasive alien species establishment. These climatic extreme events also affected the resilience of marshes and riparian ecosystems. In Mexico, the vegetation of San Jose del Cabo (estuary) is resilient to hurricanes, but the vegetation cover loss due to the increased runoff caused by stronger hurricanes generated clearings that favoured the establishment of invasive alien species such as *Arundo donax* (giant reed) and *Tamarix* sp. (tamarisk) (Shiba-Reyes *et al.*, 2021).

Extreme climatic events can also act as a driver affecting the decline of invasive alien species. For example, a heavy drought in North America between 1987 and 1988 has led to the declines of 10 alien insect herbivore species in the following few years (Ward & Masters, 2007). An extreme cold spell in southern Florida led to declines in the abundance of an alien species, *Centris nitida* (oil-collecting bee), previously introduced from Mexico and Central America (Downing *et al.*, 2016).

3.3.4.4 Carbon dioxide enrichment in air, water

The concentration of atmospheric CO_2 in 2019 was 45 per cent higher (410 ppm) than in 1750; and in part excess CO_2 released from anthropogenic sources has been taken up by the oceans, ultimately leading to decreasing pH levels (IPCC, 2021). Responses to increasing atmospheric CO_2 differ between species within terrestrial and aquatic environments.

For terrestrial plants, higher levels of CO_2 cause an increase in water use efficiency and fertilization effects that can enable greater biomass production leading to an advantage of C4 rather than C3 plants (generally benefiting native relative to alien species; Nowak *et al.*, 2004). Nevertheless, winners and losers depend on availability of nutrients and some fast-growing C3 species (such as annual grasses) may respond more strongly than slow-growing C3 herbs or C4 plants (Poorter & Navas, 2003). In arid and semiarid ecosystems invasive alien annual grasses have a competitive advantage under elevated CO_2 (Chambers *et al.*, 2014; S. D. Smith *et al.*, 2000), whereas in savannahs, native grasses may be replaced by woody alien species (Bond & Midgley, 2000; Gritti *et al.*, 2006). These performance effects can translate to increased spread of invasive alien plant species and increased CO_2 concentrations. For example, invasive alien *Phragmites* spp. (reed) benefit from higher CO_2 concentrations and as a result increased dispersal and productivity allowing these species to compensate for transpiration water loss in the coastal marshes of North America (Eller *et al.*, 2014). Similar patterns of increased performance at higher than ambient CO_2 concentrations have been found for other invasive alien plants including *Prunus laurocerasus* (cherry laurel; Hattenschwiler & Körner, 2003) and *Pueraria montana* var. *lobata* (kudzu; Forseth & Innis, 2004). Terrestrial animal responses to the rise in CO_2 will likely be indirect, based on the responses of plants, and thus are likely to be most evident for herbivorous invertebrates but will be dependent on the specific host plant, making generalizations difficult (Dukes, 2000).

One-third of the anthropogenic CO_2 has been absorbed by the oceans (J. Johnson *et al.*, 2016; Sanford *et al.*, 2014). Together with warming and altered ocean circulation, which reduce subsurface oxygen concentrations, the rising atmospheric CO_2 leads to ocean acidification (Doney *et al.*, 2012). The impacts of acidification are more pronounced in extreme regions such as in polar regions (Fabry *et al.*, 2009) and for coral reefs where calcareous corals and algae are replaced by noncalcareous algae (Hall-Spencer *et al.*, 2008). In these regions, invasive alien species that are tolerant of high CO_2 concentrations increase in abundance, as documented for macroalgal biological invasions in the northeast Atlantic (Brodie *et al.*, 2014). In the Mediterranean Sea, invasive alien genera (e.g., *Sargassum, Caulerpa* and *Asparagopsis*) spread to sites where native coralline algae are disappearing due to acidified waters (Hall-Spencer *et al.*, 2010).

3.3.4.5 Fire regime changes

In addition to direct human-induced changes to fire regimes (**section 3.3.1.5.2**; **Chapter 1**, **Box 1.4**), weather conditions that favour fire occurrence (i.e., hot, dry and windy events) have become more common in some regions due to climate change, a trend that is expected to occur in even more regions in the future (IPCC, 2021). Climate change is expected to lead to more extreme and frequent fires globally (Hoegh-Guldberg *et al.*, 2018), and there is evidence that climate change during the last decades has already increased fire activity, for example in western United States (Abatzoglou & Williams, 2016; S. E. Mueller *et al.*, 2020). Likewise, recent increases in fire frequency and severity in eastern Australia, including the unprecedented estimated area of over 10 million hectares burnt during the 2019-2020 season (Boer *et al.*, 2020; R. H. Nolan *et al.*, 2020), are consistent with predicted changes in the fire regime under climate change (Clarke & Evans, 2019; Lewis *et al.*, 2019).

Fire may facilitate establishment and spread of invasive alien plants that exhibit highly effective post-fire regeneration, and presence of these species may in turn lead to changes in fuel properties that ultimately increase fire activity, thus promoting positive feedback mechanisms to the detriment of native species (Aslan & Dickson, 2020; Brooks *et al.*, 2004; Gaertner *et al.*, 2014; Rodewald & Arcese, 2016; Serbesoff-King, 2003; **Chapter 1**, **Box 1.4**; **Chapter 4**, **Box 4.5**). Accordingly, by leading to longer fire seasons, shorter fire return intervals and/or higher fire intensity than were previously encountered, climate change may favour the establishment and spread of fire-adapted invasive alien species (Abatzoglou & Kolden, 2011).

The effect of climate change on fire regimes will likely be intensified in the future and drive the spread of *Bromus tectorum* (downy brome) in deserts and Mediterranean ecosystems in the western United States (Abatzoglou & Kolden, 2011; Balch *et al.*, 2013). Higher fire activity under climate change may also drive the spread of the African grass *Cenchrus cilliaris* (buffel grass) in the central rangelands and eastern woodlands of Australia (D. W. Butler & Fairfax, 2003; G. Miller *et al.*, 2010). The mechanism for the spread of invasive alien grasses under climate change both in the United States and Australia is an intensification of fire-invasive positive feedback loops promoted by invasive alien species in these ecosystems (i.e., grass-fire cycle), where increased production of biomass by invasive alien grasses leads to increased fire frequency, continuity and/or intensity and hence favours their spread (Balch *et al.*, 2013; D. W. Butler & Fairfax, 2003; Gaertner *et al.*, 2014; G. Miller *et al.*, 2010). In addition, fire-induced air currents associated with the recent extreme fires in Australia seem to have driven the introduction of invasive alien species to New Zealand, such as the pathogenic fungi *Austropuccinia psidii* (myrtle rust; Australian Government, 2021), and could possibly also favour the arrival of the *Agrotis infusa* (bogong moth) in the country (Warrant *et al.*, 2016).

There is evidence that increased fire activity under climate change may also directly drive the spread of invasive alien woody species. In Patagonia, a warmer and drier climate is implicated in the spread of alien pines and shrubs whose persistence is promoted by fire (Cavallero & Raffaele, 2010; K. T. Davis *et al.*, 2019; Raffaele *et al.*, 2016). In European and southern African Mediterranean ecosystems, changes in the fire regime under a warmer and drier climate change are expected to favour the spread of invasive alien tree species from the genus *Acacia* (e.g., *Acacia longifolia* (golden wattle); Souza-Alonso *et al.*, 2017).

There is very limited information on how changes in fire regimes under climate change may facilitate the spread of invasive alien animals. In freshwater ecosystems in the western United States, increased fire activity due to climate change may favour alien fishes, especially in degraded and fragmented landscapes (Dunham *et al.*, 2003). In these ecosystems, larger and more frequent fires under climate change tend to increase water temperature and decrease stream stability and connectivity, thus driving the spread of generalist alien fishes to the detriment of native Salmonidae fishes (Isaak *et al.*, 2010; Luce *et al.*, 2012), although the effect of fire as a driver affecting invasive alien species in this region may be species-specific and less significant in comparison to other anthropogenic disturbances (Sestrich *et al.*, 2011).

3.3.4.6 Sea level rise

Global mean sea level increased 0.20 m from 1901 to 2018, with an annual increase of 3.7 mm per year from 2006 to 2018 (IPCC, 2021). Sea level rise is caused by climatic factors affecting the thermal expansion of water and the melting of glaciers, permafrost and polar ice sheets (Hoegh-Guldberg & Bruno, 2010; Oppenheimer *et al.*, 2019; Rignot *et al.*, 2018). Rising sea level will likely lead to increased impacts from extreme weather events and storm surges, increased coastal flooding, higher high tides and increased saltwater intrusion into freshwater systems, altering environmental conditions along coastal zones (Nicholls *et al.*, 2014; Woodruff *et al.*, 2013). Few studies have assessed the direct effect of sea level rise as a driver in the context of biological invasions. However, it is likely that marine invasive alien species able to disperse by ocean currents (e.g., *Mytilus galloprovincialis* (Mediterranean mussel)) may be introduced to new areas due to increased inundation of coastal areas (McQuaid & Phillips, 2000). In Hawaii, the combination of sea level rise and high tide events resulted in habitat creation, facilitating the spread of invasive alien fish (e.g., tilapias) from fishponds to nearby anchialine pools (Marrack, 2016).

Sea level rise, *via* the effect of saltwater intrusion into freshwater ecosystems and increased water salinity, may also shift selection pressures and facilitate the establishment of invasive alien species. Evidence from coastal areas in the United States (K. Williams *et al.*, 1999), Australia (Traill *et al.*, 2011) and China (W. Wang *et al.*, 2015) show that sea level rise may alter soil chemistry and native vegetation patterns in coastal wetlands or native forests, selecting for species with a higher tolerance to saline habitats and where present, allow invasive alien species to dominate. Increased tolerance of the alien haplotype of *Phragmites australis* (common reed) has allowed it to spread through native salt marshes in the Atlantic and Gulf Coastal areas (Bhattarai & Cronin, 2014; Vasquez *et al.*, 2005). Similarly, *Osteopilus septentrionalis* (Cuban tree frog), an invasive alien species in the United States, is tolerant to increased salinity which can facilitate its spread and establishment in coastal environments (M. E. Brown & Walls, 2013). Verbrugge et al (2012) found that salinity tolerance of *Corbicula fluminalis* (Asian clam) and *Potamopyrgus antipodarum* (New Zealand mudsnail) was higher than for native species occurring in the River Rhine. However, models of changes in salinity due to rising sea levels in the freshwater ecosystem of the Everglades indicate a decrease in alien fish species biomass (Romañach *et al.*, 2019) compared to native species.

In coastal systems, the effect of rising sea levels on soil moisture in coastal dune systems can disrupt sediment transfer impacting dune formation processes and vegetation patterns. For example, in New Zealand, increased soil moisture disrupts the formation of dunes (Thomas *et al.*, 2018) and allows for their colonization by plants, including the invasive alien grass *Calamagrostis arenaria* (marram grass). In Australia, invasive alien *Thinopyrum junceiforme* (sea wheatgrass) is able to take advantage of increased soil salinity (Hilton *et al.*, 2006).

Sea level rise could also drive the intentional introduction of invasive alien species for the purposes of climate change adaption to reduce impacts of coastal erosion and infrastructure damage. For example, *Sporobolus* species, introduced to reduce the effects of coastal erosion (Ge *et al.*, 2015), have become widespread invasive alien species in China (An *et al.*, 2007). The introduction of *Calamagrostis arenaria* (marram grass) in Australia and New Zealand are further examples of alien species introduced for dune stabilization for coastal protection. The need for improved coastal protection from rising sea levels could see alien species being utilized in many parts of the world. Alternatively, the construction of hard surfaces for coastal protection against sea level rise can also facilitate the establishment of invasive alien species of seaweed (Geraldi *et al.*, 2014).

3.3.5 Invasive alien species

Although studying the role of invasive alien species as a direct driver of change in nature affecting invasive alien species might sound like circular reasoning, there is increasing evidence of the role that invasive alien species may play in facilitating other alien species **(Figure 3.21)**. The process by which facilitation among alien species potentially accelerates the accumulation of introduced species has gained its own term, "invasional meltdown" (Simberloff, 2006; **Chapters 1**, **2**, **4**).

3.3.5.1 Biotic facilitation

Invasive alien species can facilitate the establishment and spread of other invasive alien species through multiple direct and indirect ecological interactions **(Box 3.10)**. Direct biotic

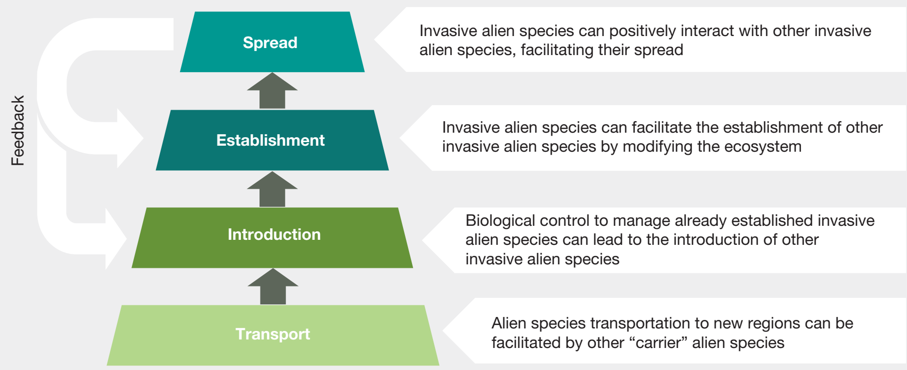


Figure 3 21 **Examples of the role of invasive alien species in facilitating additional invasive alien species across stages in the biological invasion process.**

Invasive alien species may facilitate further biological invasions by acting as pathways for other invasive alien species, by motivating the deliberate introduction of additional alien species for biological control, by modifying the ecosystem thereby facilitating the establishment other alien species, or by facilitating spread. Biological invasions can also negatively impact ecosystem functioning and resistance, causing feedback *via* facilitation of the introduction and establishment of additional invasive alien species.

facilitation often involves plant-animal interactions (e.g., pollination; e.g., Morales & Aizen, 2002), dispersal, (e.g., Mandon-Dalger *et al.*, 2004), plant-fungal interactions (e.g., mycorrhizal symbiosis; Dickie *et al.*, 2010), or animal-animal (e.g., ant-scale insect) mutualisms (Richardson, Allsopp, *et al.*, 2007; Simberloff & Von Holle, 1999; Traveset & Richardson, 2014), or even multitrophic interactions like those between alien herbivores dispersing alien fungi (e.g., mycorrhiza of alien trees; Nuñez *et al.*, 2013). However, alien species may also indirectly facilitate the establishment and spread of other invasive alien species, by modifying the biotic conditions of the recipient community (e.g., reducing competition or predation pressure by a third species, or increasing food resource availability), or abiotic attributes and ecosystem properties (e.g., by promoting habitat disturbance such as enhanced fire regimes; see **section 3.3.4.5**) increasing soil nutrients by nitrogen-fixing plants, etc.

The term "invasional meltdown" coined by Simberloff & Von Holle (1999), refers to the process by which alien species facilitate one another, magnifying ecological effects, leading to accelerating rates in the number of invasive alien species and magnification of impacts. In other words, invasional meltdown is the potential emergent result of a series of facilitations (Ricciardi, 2001). Therefore, although invasional meltdown has often been broadly used to refer to any kind of positive interaction among alien species in the peer-reviewed literature, this chapter refers to invasional meltdown *sensu* (Simberloff & Von Holle, 1999), and shows how this phenomenon is linked to facilitation among invasive alien species as a driver that may accelerate rates of biological invasions (**Chapter 2**), and synergistic impacts (**Chapter 4**).

A recent review based on 150 empirical studies, Braga *et al.* (2018), confirmed overall broad support for facilitative interactions among alien species (63.3 per cent of the studies) across multiple types of interactions (direct or indirect, unidirectional, reciprocal, or multi-species), ecological levels (individual, population, community and ecosystem), taxonomic groups and major habitat types. This evidence points to biotic facilitation among alien species as a major driver facilitating the establishment and spread of invasive alien species. However, they also found some exceptions to this general pattern, and have identified biases and gaps (see below). This section reviews the role of biotic facilitation among alien species as a driver affecting the different stages of the invasion process (Braga, Gómez-Aparicio, *et al.*, 2018; Gavira-O'Neill *et al.*, 2018; Jeschke *et al.*, 2012).

The transportation of alien species to new regions can be facilitated by other "carrier" alien species. For instance, plants relying on endozoochoric or ectozoochoric seed dispersal can be transported and introduced to new regions in the guts, fur, hoof or feathers of alien animals (Reynolds *et al.*, 2015; Van Leeuwen, 2018; Diaz Velez *et al.*, 2020). The alien mammal *Axis porcinus* (hog deer) disperses similar numbers of alien and native plant species' seeds through its faeces, thus greatly facilitating the dispersal of alien species in south-eastern Australia (N. E. Davis *et al.*, 2010), and horses transport alien seeds on their hooves (Gower, 2008). Indeed, facilitation is concomitant

Box 3 10 Three-way invasional meltdown: invasive alien ungulates disperse invasive alien fungi that facilitate pine invasions.

One example of invasional meltdown that involves belowground mechanisms of facilitation, is that of alien ungulates dispersing alien ectomycorrhizal fungi, in turn facilitating the invasion by alien pine trees **(Figure 3.22)**. Introduced pine trees have become invasive in many parts of the southern hemisphere and cause profound ecological, social and economic impacts. Pine tree establishment and growth are critically dependent on the interaction with ectomycorrhizal fungi, which provide nutrients, water and protection against pathogens, in exchange for plant carbon. Pine trees thus co-invade with alien ectomycorrhizal fungi. As ectomycorrhizal fungi disperse independently from pine trees, some ectomycorrhizal fungi species are able to disperse away from the original place of introduction, establish a spore bank, and make stands of native species more susceptible to pine invasions. In turn, invasive alien ungulates consume alien ectomycorrhizal fungal sporocarps and disperse the spores of some ectomycorrhizal fungi species through their faeces (Nuñez *et al.*, 2013). This mechanism of dispersal is crucial for the pine-ectomycorrhiza symbiosis both in the alien and native ranges fungi, especially for those ectomycorrhizal fungi that produce hypogenous sporocarps (i.e., truffle-like fungi), which are proposed to exclusively rely on mammal-mediated dispersal **(Figure 3.22)**. Invasive ungulates can disperse viable spores in high densities, far beyond the distance they typically disperse through wind (Horton, 2017). This scale of dispersal is important considering the scale at which biological invasion occurs. Although not all ectomycorrhizal fungi species survive this form of dispersal, evidence shows that those ectomycorrhizal fungi species that survive or even depend upon mammal-mediated dispersal are among the most invasive ectomycorrhizal fungi (Policelli *et al.*, 2019). Dispersal limitation of ectomycorrhizal fungi might be determinant for invasions of Pinaceae. The absence of dispersal vectors, such as ungulates, could act as an impediment for viable ectomycorrhizal fungi propagules to reach sites far from the propagule source, in turn hindering the invasion by the alien plant host or increasing its lag time.

Box 3 10

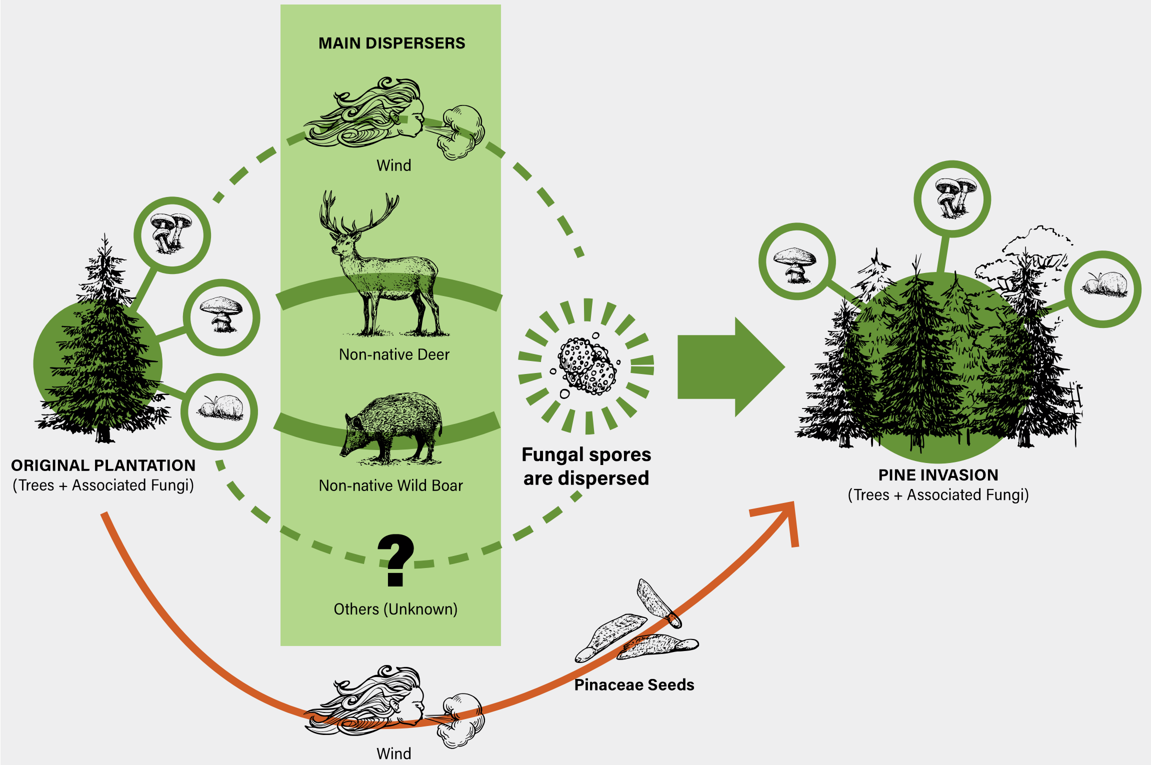


Figure 3 22 **Diagram of the three-way invasional meltdown between invasive alien pine trees, invasive alien ectomycorrhizal fungi and invasive alien ungulates.**

Pine trees need ectomycorrhizal fungi to successfully invade (thick middle arrow). Pine seeds are mainly dispersed by wind (bottom arrow). Spores from ectomycorrhizal fungi can be dispersed through wind (upper dotted line) but some ectomycorrhizal fungi species present in the pine plantation produce sporocarps that are eaten by invasive alien ungulates (wild boar and deer; medium thick lines). Ungulates transport the invasive ectomycorrhizal fungi spores further from the invasion source population compared to wind. Spores from some invasive ectomycorrhizal fungi can form long-lasting spore banks in the soil, making native stands more susceptible to pine invasion. More evidence is needed about other potential mechanisms of fungal dispersal (here represented by the question mark, lower dotted line), such as bird dispersal, or human dispersal which could also be important, especially over long-distances. Source: Policelli *et al.* (2022), under license CC BY 4.0. https://doi.org/10.1007/978-3-031-12994-0_2

with the uptake, transportation and introduction of invasive alien species that engage in obligated symbiosis with other organisms and in the co-introduction of alien parasites by their alien hosts (e.g., Arbetman *et al.*, 2013). As an example, frugivorous birds have been shown to simultaneously disperse three interlinked alien species: the seeds of *Ligustrum lucidum* (broad-leaf privet), its weevil granivore and a parasite of the weevil (Chen *et al.*, 2021). However, despite the importance of transport and introduction within biotic facilitation **(Figure 3.4)**, these initial two stages of the biological invasion process have been largely neglected in major studies and reviews about facilitation among invasive alien species or invasional meltdown (O'Loughlin & Green, 2017; Simberloff, 2006; Simberloff & Von Holle, 1999).

Once alien species have arrived in a new region, positive interactions with other alien species may be unidirectional or bidirectional. Unidirectional interactions include alien species facilitating any aspect of another's survival, reproduction, resource acquisition, or other factor that enhances establishment, population growth, or spread while the latter has no detectable influence on the former (a commensal relationship). With bidirectional interactions, both species have a reciprocal positive effect (mutualism). Multispecies interactions may be through direct and/or indirect effects (reviewed in (Braga, Gómez-Aparicio, *et al.*, 2018). According to Braga *et al.* (2018), most studies focused on unidirectional or multi-species interactions (58 studies each) and these generally found a high level of support for facilitation, while there were fewer studies on reciprocal interactions (34 studies) and these generally found a lower level of support for facilitation (Braga, Gómez-Aparicio, *et al.*, 2018). As an additional scenario, one alien species could facilitate the success of another at its own expense, such as through a predator-prey or parasite-host relationship; such exploitations are likely quite common, but are also rarely considered in studies examining facilitations (but see Grosholz, 2005; Ricciardi, 2001). Empirical evidence exists for multispecies interactions that affect the success of an invasive alien species or trigger the expansion of an alien species. For example, Grosholz (2005) showed that the invasion by a predator, *Carcinus maenas* (European shore crab), reduced the abundance of a native clam, which was the crab's preferred prey, by 10-fold,; this interaction released another introduced species, *Gemma gemma* (amethyst gemclam), from competition and thus allowed it to become superabundant, after having been present at low abundance for decades. However, while many studies have inferred invasional meltdown, few cases have demonstrated an accelerating rate of establishment or spread of invasive alien species and/or the synergistic impact of these invasive alien species. Heimpel *et al.* (2010) described a scenario of invasional meltdown where *Aphis glycines* (soybean aphid) increased the abundance of eleven invasive alien species including worms, shrubs, birds, beetles and animal and plant pathogens. A study by Ricciardi (2001) of the North American Great Lakes found that facilitative interactions among alien species were at least as common as antagonistic interactions, and that the rapid accumulation and synergistic effects of alien species, while best explained by increased propagule pressure (e.g., from shipping), was consistent with the prediction of the invasional meltdown hypothesis (Simberloff & Von Holle, 1999). Christmas Island, mentioned previously, provides the best documented case of invasional meltdown to date. After being introduced to the island several decades ago, *Anoplolepis gracilipes* (yellow crazy ant), persisted for decades at low density before its population exploded in the late 1980s. The ants had an antagonistic relationship with *Gecarcoidea natalis* (Christmas Island red crab), a keystone omnivore. The ants caused a reduction in populations of the crab, which resulted in increased tree seedling density and reduced leaf litter on the forest floor. Further, the depletion of *Gecarcoidea natalis* promoted yet another invasive alien species, *Lissachatina fulica* (giant African land snail). Simultaneously, in the forest canopy, the higher density of ants promoted population growth of introduced honeydew-secreting scale insects through a mutualistic relationship, which resulted in fungal growth and dieback of trees. This ecosystem transformation occurred in a period of only a few years (Green *et al.*, 2011; O'Dowd *et al.*, 2003).

Finally, Braga *et al.* (2018) identified biases in research effort. In particular, the majority of the studies focused on the individual and population levels (about 44 per cent each), with a lower representation of studies at the community (10.5 per cent) and ecosystem levels (1.5 per cent), and there are less studies addressing indirect effects (56 studies) than direct effects (87 studies). Regarding habitats, most evidence comes from terrestrial ecosystems (63.1 per cent) compared to fresh and marine ecosystems (21.5 and 15.4 per cent, respectively). As for taxonomic groups, more studies focused on plants and algae (89 studies), followed by invertebrates (83 studies) and vertebrates (51 studies). In the future, taking the importance and prevalence of alien-alien facilitation into account might lead to better prediction of the outcomes of biological invasions and effective prevention (**Chapter 5**). Moreover, while there have been many examples of invasive alien species facilitating one another, suggesting that invasional meltdown is possible in a broad range of ecosystems, there is thus far very little published evidence of an accelerated accumulation of invasive alien species attributable to these facilitations (but see O'Dowd *et al.*, 2003; Ricciardi, 2001; Simberloff & Von Holle, 1999).

3.3.5.2. Unintended consequences of management through biological control

Many empirical examples of the unintended consequences of management of biological invasions resulting in the introduction, establishment or spread of invasive alien species stem from the literature on early attempts at biological control (**Chapter 5**, **section 5.6.2.3**). Many of these historical high-profile cases report negative direct impacts on non-target native species by generalist predators or pathogens released as biological control agents (e.g., the release of cats and mongoose to control rodents, cane toads against agricultural pests and plant pests or diseases to control invasive plants). These examples are all from a time when biological control was implemented in an unregulated way, for example, with no requirement for risk assessment of the biological control agent (**Chapter 5**). A classic example is *Cactoblastis cactorum* (cactus moth) that was intentionally released on islands in the Caribbean in 1957 for the control of native *Opuntia* (pricklypear) species that were seen as a nuisance to tourists. However, the moth spread from the Dominican Republic to Florida, where it poses a threat to native *Opuntia* (Hinz *et al.*, 2019), and to the Yucatán peninsula where it was successfully eradicated (Senasica, 2019). It has currently spread across the south-eastern United States to Texas where it can enter Mexico again and threaten the over 100 native *Opuntia* species, many of which are endemic and constitute an important part of the Mexican diet and economy (Senasica, 2019).

Insects have also been released to control other insects and in some cases these biological control agents have become invasive alien species. Examples include two species of ladybird, *Harmonia axyridis* (harlequin ladybird) from Asia causing declines of native ladybirds in the United Kingdom and Belgium (Roy *et al.*, 2012) and in other countries worldwide (Roy *et al.*, 2012) and *Coccinella septempunctata* (seven-spot ladybird) from Europe impacting populations of native North American ladybirds (E. W. Evans *et al.*, 2011). The release of *Euglandina rosea* (rosy wolf snail) in Hawaii to control *Lissachatina fulica* (giant African land snail) also failed to lead to desired management outcomes but instead resulted in the intentional introduction of an invasive alien species (Cowie, 2001).

Rhinella marina (cane toad) was introduced into Queensland, Australia in 1935 from Hawaii to control insect pests of sugar cane and became an invasive alien species with direct and possible indirect impacts on non-target native Australian vertebrates (R. Shine, 2010). This is an example where science was ignored and a political decision was made to introduce *Rhinella marina* (M. D. Day *et al.*, 2021). Mammals have also been released as biological control agents with unintended consequences. *Herpestes javanicus auropunctatus* (small Indian mongoose) was introduced initially to control alien invasive rodents and snakes in the West Indies, Hawaiian islands and Japan, and has resulted in mammal and reptile extinctions (Hays & Conant, 2007). In aquatic environments, *Gambusia affinis* (western mosquitofish) and *Gambusia holbrooki* (eastern mosquitofish), native to the fresh waters of the United States, have been introduced worldwide as biological control agents of mosquito larvae, but are implicated in the decrease and loss of non-target native invertebrates as well as fish and amphibian populations (Azevedo-Santos *et al.*, 2017). In the marine realm, the sea urchin *Evechinus chloroticus* (kina) released as an augmentative biological control agent against the invasive *Undaria pinnatifida* (Asian kelp) showed substantial non-target effects on benthic communities, however these were localized and reversible (Atalah *et al.*, 2013). Conservation actions may also have unintended consequences in facilitating invasive alien species, as illustrated by the creation of beaver dams during the reintroduction of *Castor canadensis* (North American beaver) in Verde River, Arizona, United States, which shifted desert fish assemblages toward dominance by alien species (P. P. Gibson *et al.*, 2015; Reaser, 2003).

A global review of non-target impacts of weed biological control agents found that the proportion of intentionally released biological control agents causing non-target effects declined from 18.2 per cent prior to the 1960s to 9.9 per cent in the period 1991–2008 (Hinz *et al.*, 2019). Similarly, an analysis of all non-target effects of weed biological control programmes from 1969 to 2014 showed a risk factor of less than 1 per cent (Moran & Hoffmann, 2015). There was no evidence of non-target impacts from plant pathogens used for biological control (Suckling & Sforza, 2014).

3.4 ADDITIONAL DIRECT DRIVERS – NATURAL DRIVERS AND BIODIVERSITY LOSS

3.4.1 Natural hazards

Natural large-scale disturbances, such as hurricanes, earthquakes and tsunamis can facilitate the onward spread of alien species from an existing invaded range to new regions (Carlton *et al.*, 2017; **section 3.3.3.3**, **Box 3.8**) as well as encourage their wider spread within regions where they are already present as aliens (Bellingham *et al.*, 2005; **Figure 3.23**). Natural drivers have thus facilitated the wider establishment and spread of alien plants and animals within and beyond their known invaded range through acting as agents of secondary dispersal (e.g., Lovette *et al.*, 1999; Toepfer, 2012; Lee *et al.*, 2014; Massa *et al.*, 2014).

The roles of natural drivers apply to all regions and all realms. Natural disturbances such as hurricanes (**section 3.3.4.3**) have played a role in assisting the dispersal of alien animals (Andraca-Gómez *et al.*, 2015; Censky *et al.*, 1998; Johnston & Purkis, 2015), plants (Bhattarai & Cronin, 2014) and microbes (Feehan *et al.*, 2016), leading to expansion of their historical invaded ranges. A classic example of onwards dispersal of alien species *via* natural drivers is the crossing from Africa to the Americas by the *Bubulcus ibis* (cattle egret), whose introduction, establishment and further spread throughout the Americas has been linked to multiple weather events (Massa *et al.*, 2014). Wind and ocean currents offer new opportunities for colonization within and beyond the invaded range of both marine and terrestrial organisms (Munoz *et al.*, 2004); rafting during extreme weather events is a common example by which non-flying alien animals can be transported between islands, as has been documented for *Iguana iguana* (iguana; Censky *et al.*, 1998). As species' thermal barriers are being altered or lifted by climate change, ocean currents are contributing to the range expansion of alien species and colonization of previously inhospitable regions such as the Arctic (Chan *et al.*, 2019) and Antarctica (C. I. Fraser *et al.*, 2018). Disease outbreaks among *Strongylocentrotus droebachiensis* (green sea urchin) in the northwest Atlantic Ocean have been attributed to *Paramoeba invadens*, a pathogenic amoeba that is intolerant of the typical winter sea surface temperatures in the region. Evidence suggests that the amoeba originates in southern surface waters transported to the north Atlantic coast, and that disease outbreaks have occurred during hurricanes and unusual warm winter sea temperatures (Feehan *et al.*, 2016). Molecular and oceanographic evidence suggests that ocean currents regularly disperse rafting species thousands of kilometres, and that Southern Ocean coasts are biologically well-connected (C. I. Fraser *et al.*, 2022). If warm-adapted

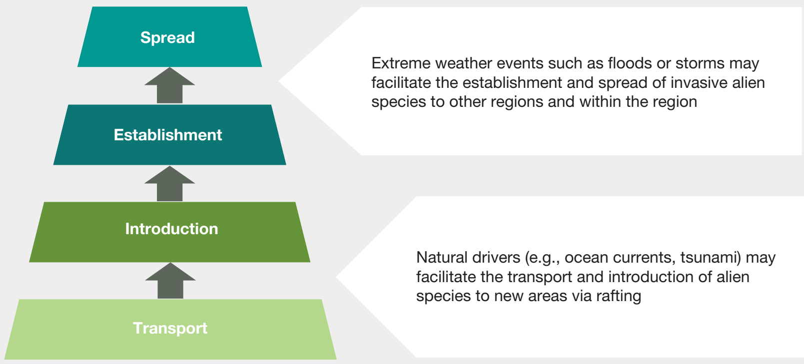


Figure 3 23 **Examples of the role of natural hazards in facilitating invasive alien species across stages of the biological invasion process.**

Natural hazards can facilitate biological invasions by acting as agents of secondary dispersal beyond and within the invaded range. Examples are chosen to represent relationships found across the invasion stages.

taxa frequently disperse to Antarctic waters, global warming could allow the region to become increasingly colonized by new species delivered *via* ocean rafting, especially during storms (C. I. Fraser *et al.*, 2018). These cases illustrate the capacity of natural drivers to facilitate colonization events, but also suggest an increasing influence of anthropogenic climate change as an amplifier of such dispersal opportunities, by, for example, altering or strengthening ocean currents, or by creating temporary hydrological connections (**section 3.3.4**).

The relative importance of natural drivers such as natural hazards in the range dynamics of alien species, however, are likely to be relatively minor. Natural long-distance dispersal events have evidently been sufficient to colonize remote oceanic islands in the prehistoric past, but are likely orders of magnitude less frequent than human-assisted, long-distance dispersal events (Ricciardi, 2007). Natural hazards typically move small numbers of propagules and are dependent on weather patterns and other environmental constraints. This is in contrast to modern human-assisted biological invasions in which enormous numbers of individual organisms and a broad diversity of species can be moved to virtually any region of the planet over short time scales (Ricciardi, 2007). The spread of alien species often involves a combination of human-assisted and natural drivers, of which the latter may be dominant at small spatial scales (Chan *et al.*, 2019; Medley *et al.*, 2015).

3.4.2 Biodiversity loss and ecosystem resilience

It is estimated that 25 per cent of all species globally are threatened by extinction and that 1 million species may become extinct in the following decades due to human interference, especially land- and sea-use change (**section 3.3.1**) and direct exploitation of natural resources (**section 3.3.2**) (IPBES, 2019). While this biodiversity loss has been the dependent variable in most previous IPBES assessments (IPBES, 2019), for invasive alien species it can also be seen as a driver that may facilitate biological invasions since reduced taxonomic or functional diversity of native ecosystems may reduce their biotic resistance and thereby facilitate the establishment and spread of invasive alien species (Levine *et al.*, 2004; **Figures 3.3** and **3.24**). A wide range of biotic interactions may confer resistance to biological invasions in native communities, including competition, predation, herbivory and disease, and all of these may be involved in constraining the introduction, establishment and spread of invasive alien species (Alofs & Jackson, 2014; Elton, 1958; Levine *et al.*, 2004).

Studies from terrestrial systems show that the presence and diversity of native vegetation, along with phylogenetic distance between the invasive alien species and resident community, can constrain plant invasions, implicating different modes of plant-plant competition for space as a powerful mechanism underlying biotic resistance to invasive alien species. For example, Byun *et al.* (2015) analyzed the interplay between abiotic constraints, propagule pressure and biotic resistance by conducting experiments to explore the *Phragmites australis* (common reed) invasion process, and found that maintaining native plant cover could confer invasion resistance, even when abiotic conditions changed (Byun *et al.*, 2015). Going *et al.* (2009) found that competition from a resident annual plant community had strong negative effects on the biomass and reproduction of invasive alien grass, such as *Avena barbata* (slender oat), *Bromus diandrus* (great brome) and *Hordeum murinum*

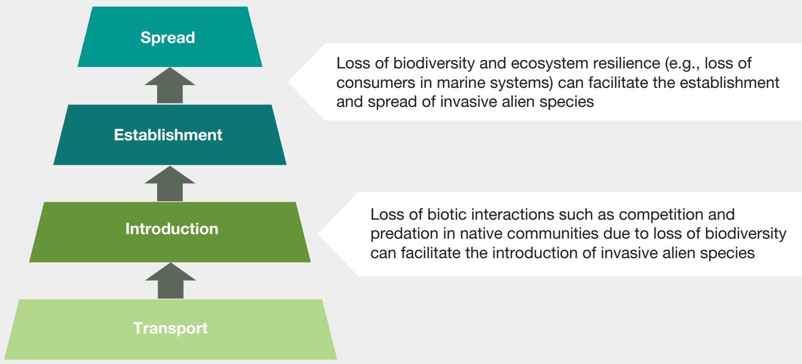


Figure 3 24 **Examples of the role of biodiversity loss in facilitating invasive alien species across stages of the biological invasion process.**

Biodiversity loss can facilitate biological invasions by reducing ecosystem resistance to invasion, which may facilitate the introduction, establishment and spread of invasive alien species.

(mouse barley), and that removing the resident communities increased the biomass and seed production of the invasive alien species by two-fold to ninefold. Elton's diversity-invasibility hypothesis proposes that taxonomic diversity in native communities confers additional resistance to invasive alien species because less niche space is available to alien species (Elton, 1958). In support of this hypothesis, Maron & Marler (2007) found that plant assemblages with higher plant species richness displayed lower invasibility (**Chapter 1**, **section 1.3.2**) than assemblages with lower species richness, and Zheng *et al.* (2018) found that the invasion success of *Chromolaena odorata* (Siam weed) correlated negatively with both biomass and species richness of the native community. A third idea related to biotic resistance is Darwin's naturalization hypothesis, which builds on Darwin's (1859) position that alien species will be more successful in a native community if they are more distantly related to native residents, because relatedness may indicate niche similarity (**Chapter 1**, **section 1.3.2**; Violle *et al.*, 2011). In support of this hypothesis, Zheng *et al.* (2018) experimentally established that success of the invasive tropical shrub *Chromolaena odorata* increased with functional distance to the native community. Similarly, Iannone *et al.* (2016) found, in a large-scale observational study of invasive alien species in forests across the eastern United States, that tree biomass and evolutionary diversity, but not species richness, was negatively associated with the establishment and dominance of the invasive alien species, and thus that evolutionary diversity is indicative of biotic resistance. In an experimental study explicitly designed to distinguish Darwin and Elton's hypotheses, Feng *et al.* (2019) found support for both ideas, as the effects of both phylogenetic and functional distance became stronger as species richness increased, and further analyses indicated that both competitive inequalities and niche differences between invasive alien and native communities may contribute to these responses.

Observed relationships between the taxonomic or functional diversity of native and invasive alien plants may be caused by mechanisms beyond direct plant-plant competition. For example, observed biotic resistance from native vegetation against invasive alien plant species may operate *via* soil pathogens that negatively affect the invasive alien species (Knevel *et al.*, 2004; van Ruijven *et al.*, 2003). Similarly, a study from south-western United States by St. Clair *et al.* (2016), found that native rodents suppressed invasion by *Bromus tectorum* (downy brome) while promoting native plant diversity after fire, providing strong biotic resistance to invasive alien plants through preferential seed and seedling predation on invasive alien species.

A meta-analysis of marine experiments revealed the same general trend of diversity-mediated biotic resistance as was observed in terrestrial systems; high native primary producer diversity in marine systems confers significant resistance to alien primary producers through competition, whereas low-diversity communities in the same marine systems often fail to do so (Kimbro *et al.*, 2013). However, unlike terrestrial systems, biotic resistance in freshwater and marine environments might to a larger extent be driven by consumption (Alofs & Jackson, 2014). Resident species at the top of the food chain can prevent invasion by alien species which are lower in the food chain. In freshwater systems in the southern United States, Parker & Hay (2005) found that native consumers (including crayfish, grasshoppers and slugs) preferred alien plants as a food source over native plants, conferring resistance to biological invasion. In another example, in China the native crab *Helice tientsinensis* effectively inhibits *Sporobolus* *alterniflorus* (smooth cordgrass) invasion in anthropogenic ditches, high marshes and estuarine mangrove forest by grazing seedlings and suppressing their density and survival (Ning *et al.*, 2019). In rainforest of Christmas Island, Indian Ocean, *Gecarcoidea natalis* (Christmas Island red crab) can kill the introduced *Lissachatina fulica* (giant African land snail), restricting the distribution of giant African land snails and thus conferring biotic resistance to biological invasions in undisturbed habitats on the island (Lake & O'Dowd, 1991).

3.5 MULTIPLE, ADDITIVE OR INTERACTING EFFECTS OF DRIVERS AFFECTING INVASIVE ALIEN SPECIES

The evidence that the majority of the Earth's ecosystems are subject to complex threats from several concurrent, interacting and exacerbating drivers of change in nature is unequivocal (IPBES, 2019). The net effect of drivers is often not additive, as drivers can reinforce or mitigate each other's effects (i.e., be synergistic or antagonistic; Fontúrbel, 2020; Jackson *et al.*, 2016; Pyšek *et al.*, 2020). Generalizing from studies of single drivers in isolation may therefore yield misleading conclusions (Bowler *et al.*, 2020). Interactions between multiple drivers of change are expected to jeopardize ecosystem functioning and biodiversity, and are a key concern for conservation and management (Cote *et al.*, 2016). As a special case, novel ecosystems (**Chapter 1**, **Box 1.5**), which contain new species combinations and/or altered ecosystem functioning (Hobbs *et al.*, 2006, 2009; Morse *et al.*, 2014; Seastedt *et al.*, 2008) can be more susceptible to biological invasions than more native ecosystems (Ogutu-Ohwayo *et al.*, 2016).

Associations amongst invasive alien species and other drivers of change in nature are generally understudied, and the degree to which biological invasions are affected by additive or multiplicative processes and interactions among drivers is therefore difficult to assess (**Figure 3.25**, **Chapter 1**, **section 1.3.3**). A recent review found that only 16 per cent of published research on invasive alien species examined associations with at least one other direct or indirect driver of change in nature, and less than 3 per cent considered associations with two or more additional drivers (Hulme, 2022). This section acknowledges the limited information on interactions between invasive alien species and other drivers of change in nature, and refers to a number of illustrative, rather than exhaustive, examples to highlight the importance of taking such interactions into account **(Box 3.11)**.

Indigenous Peoples and local communities recognize that most drivers of change on their lands do not act in

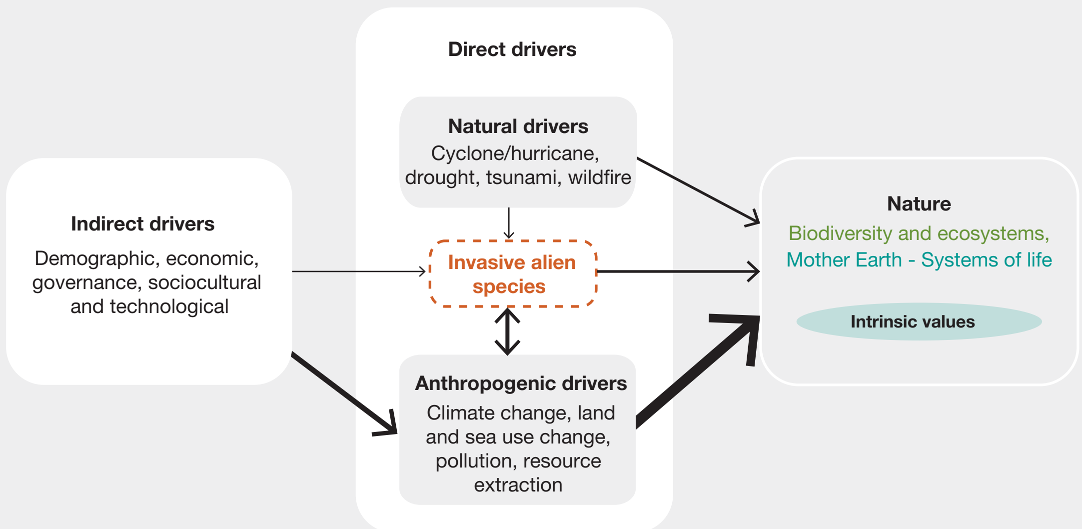


Figure 3 25 **Schematic representation of the links between indirect and direct drivers of change in nature in relation to their potential effect on invasive alien species (see Figure 3.3).**

The size of the arrows represents the available evidence in the scientific literature for the importance of the different links, as assessed though a bibliometric analysis of the literature (the absolute number of articles retrieved is reflected in the size of each arrow). Adapted from Hulme (2022), https://doi.org/10.1111/cobi.13817, under license CC BY 4.0.

Box 3 11 **Multiple interacting drivers trigger plant invasions in mountains.**

In the last two decades, evidence has highlighted the increasing number of alien plants establishing in mountain regions, despite mountains across the world differing greatly in terms of their biodiversity, climate, geology, land-use and other socioecological factors (e.g., T. Becker *et al.*, 2005; Dainese *et al.*, 2014; Guo, Fei, *et al.*, 2018; Khuroo *et al.*, 2007; Kueffer *et al.*, 2013; Marini *et al.*, 2012). In a standardized survey along elevational gradients in nine regions on four continents, more than 300 alien plant species were observed (Haider *et al.*, 2018).

Historically, agriculture and domestic grazing were probably the first and most extensive drivers that facilitated plant invasions in mountains. For example, species typical for European grasslands are widespread in the alien flora of mountains worldwide (McDougall *et al.*, 2011). However, as mountains have become more heavily developed, drivers such as infrastructure and anthropogenic land-use changes (including urbanization and the development of corridors such as roads, trails and railways) have synergistically supported the upslope movement of alien plants introduced at low and mid-elevations (Alexander *et al.*, 2011; Lembrechts *et al.*, 2017; Liedtke *et al.*, 2020; Rashid *et al.*, 2021; Yang *et al.*, 2018). Corridors provide a conduit for rapid movement of propagules aided by the continuous movement of vehicles and construction material (Rew *et al.*, 2018). Furthermore, the typically ruderal alien species benefit from the disturbed habitat conditions resulting from roadside construction and maintenance (Lembrechts *et al.*, 2016; Pickering & Hill, 2007; Seipel *et al.*, 2012).

In recent decades, and because of technological advances and higher economic and development pressures, mountain ecosystems have experienced a new phase of extensive land-use changes and infrastructure development, which has triggered the spread of invasive alien plants to even higher and more remote areas (Kalwij *et al.*, 2015; Rew *et al.*, 2018). Tourism in mountain areas has exponentially increased across the world (reviewed by Río-Rama *et al.*, 2019), increasing the development pressure in mountainous and alpine ecosystems. In most regions, there has been an increase in human settlements including housing and urbanization for tourism and recreation, as well as the expansion of infrastructure such as roads, railroads, powerlines and telecommunication towers. Overall, increasing land development, traffic and visitation rates have multiplied the chances for the introduction and establishment of invasive plant propagules at high elevations (McDougall *et al.*, 2011).

Figure 3 26 **Monte Baldo hosts an increasing number of invasive alien species.**

An increasing number of alien plants have been establishing in mountains like Monte Baldo throughout the European Alps (Dainese *et al.*, 2014). Photo credit: Katzwiekatzkann, WM Commons – under license CC BY 3.0.

Box 3 11

Climate change is expected to facilitate the expansion of invasive alien species to higher elevations both through direct and indirect effects. Climate change will reduce climatic barriers for generalist alien plants (Pauchard *et al.*, 2016), especially in regions which are not water-limited. Range shifts towards higher elevations have been reported, and alien plants appear to be moving up in elevation faster than native species (Dainese *et al.*, 2017). Increased disturbance due to, for example, higher fire frequency and intensity and insect and pathogen outbreaks triggered by climate change, will also play a role in promoting invasive alien species, especially in the middle elevations of mountains where forests will be more prone to invasion by woody plants (Franzese & Raffaele, 2017; Jactel *et al.*, 2020; Liebhold *et al.*, 2017). By reducing snow cover, climate change will promote the displacement of mountain ski facilities and resorts to higher elevations and into previously undeveloped areas. In addition, summer use of high elevation mountain resorts may be boosted in search for cooler places or "last chance tourism" (e.g., Kilungu *et al.*, 2019). The interactive effects of multiple drivers of change in nature affecting plant invasions are generally underestimated and primarily focus on climate change. However, interactions between climate change, infrastructure development, social values and land-use change will be informative because simple projections based solely on climate will be unreliable (Dainese *et al.*, 2017).

isolation in facilitating invasive alien species.[8] For instance, the interaction between economic and sociocultural drivers has been identified by Indigenous Peoples and local communities as responsible for the introduction and establishment of invasive alien species, as exemplified by the complexity and variability of societal and ecological processes facilitating the spread and establishment of *Prosopis juliflora* (mesquite; **Box 3.6**). Indigenous Peoples and local communities from Botswana, Ethiopia, Jordan and Kenya report that the main cause of the dispersal and spread of *Prosopis juliflora* is wildlife and livestock (Al-Assaf *et al.*, 2020; Bekele *et al.*, 2018; Haregeweyn *et al.*, 2013; IPBES, 2020; Mosweu *et al.*, 2013; Wakie *et al.*, 2016), which are linked to both sociocultural and economic drivers (pastoralist livelihoods). The Afar from Ethiopia now also use *Prosopis juliflora* as fuel wood, animal fodder and construction materials (Haregeweyn *et al.*, 2013) for economic reasons, which also contribute to its spread and establishment. In the Ramnad area of Tamil Nadu, India, sociocultural and land-use change led local communities to adopt charcoal-making, which promoted the spread of *Prosopis juliflora* through the reduction of grazing land and livestock holdings (Chandrasekaran & Swamy, 2016). Similar observations have been made for other species. For example, Indigenous Peoples and local communities in Ghana, West Africa and Himalayan India have observed the invasion of *Chromolaena odorata* (Siam weed; Amanor, 1991). Most of the Indigenous Peoples and local communities identify multiple drivers, such as the movement of humans and machinery, trade, land-use change, infrastructure development such as road construction and tourism, as important in facilitating the spread of this invasive alien species on their lands (Amanor, 1991; Braimah & Timbilla, 2002; Kosaka *et al.*, 2010; Timbilla & Braimah, 1993; Uyi & Igbinosa, 2013).

8. Data management report available at: https://doi.org/10.5281/zenodo.5760266

3.5.1 Land-use change and climate change

Most of the current knowledge on interactive effects of climate change and land-use change on biological invasions is informed by modelling studies designed to assess potential changes in species' distribution, with few experimental studies examining mechanisms based on demographic responses (e.g., L. C. Ross *et al.*, 2008). Modelling studies suggest that the relative importance of land-use change and climate change in facilitating biological invasions is highly variable and often species-specific or scale dependent (Febbraro *et al.*, 2019; Manzoor *et al.*, 2021). A general insight emerging from this work is that incorporating land-use change scenarios into climate change models can considerably alter the predicted outcomes of future biological invasions.

Climate change acts over broad regional and temporal scales, whereas land-use changes can have a much more local and immediate effect in response to commercial or land management decisions. For example, at the global scale, the future distribution of two major invasive alien plant species depended primarily on how their niches responded to climatic changes, with land-use change having a minor effect on distribution at this scale (Gong *et al.*, 2020). However, at smaller scales, several studies show that while future changes in temperature or precipitation patterns will exert a large influence on the establishment and spread of invasive alien plants, the rate of spread is often limited by the availability of suitable habitat (Taylor *et al.*, 2012), such as either cultivated or grazed land (J. V. Murray *et al.*, 2012; L. C. Ross *et al.*, 2008), urban areas (Nobis *et al.*, 2009) or the expansion of native ecosystems (Manzoor *et al.*, 2021). Interestingly, in some cases, the response of native vegetation to future climatic changes can enhance biological invasions. For example, a study from Wales found that the invasive forest understory shrub *Rhododendron ponticum* (rhododendron) is likely to decline under most climate and land-use scenarios, largely due to declines in coniferous forest cover, but is projected to increase under a scenario where ecosystem conservation leads to a substantial increase in coniferous forest cover (Manzoor *et al.*, 2021).

Increased fire activity may act synergistically with other climatic and land-use changes to drive the establishment and spread of invasive alien plant species (**sections 3.3.1.5.2**, **3.3.4**). For example, interactions among increased temperature, decreased precipitation and more frequent fires in recent decades have driven an upward spread of fire-adapted C4 invasive alien grasses (e.g., *Melinis minutiflora* (molasses grass)) along an elevation gradient in Hawaii (Angelo & Daehler, 2013). Higher atmospheric CO_2 levels have increased the productivity of invasive alien annual grasses in the western United States (Ziska *et al.*, 2005) and are predicted to favour the post-fire growth of African invasive alien grasses to the detriment of Australian native grasses (Tooth & Leishman, 2014). In the future, warmer minimum temperatures and other climatic changes in the western United States may also favour the establishment and spread of fire-adapted invasive alien grasses in previously unsuitable sites (Abatzoglou & Kolden, 2011; Martin *et al.*, 2015), although contractions of suitable habitat may also occur due to more extreme drought conditions (Albuquerque *et al.*, 2019). There is also concern that the interaction of climate change, fire activity and land management may promote grass-fire cycles and hence the spread of fire-adapted invasive alien grasses in disturbed forest ecosystems (Kerns *et al.*, 2020). Likewise, climate change, fragmentation and increased fire frequency have been shown to act synergistically to drive the spread of *Lantana camara* (lantana) in temperate forests in the Western Himalaya (Mungi *et al.*, 2018).

Few studies have investigated the interactions between land-use changes and climate change for invasive alien animals, invertebrates and microorganisms. Yet, the findings of existing studies often describe a prevalence of non-additive effects, reflecting those described above for plants. In an example from Italy, Febbraro *et al.* (2019), found that the potential distributions of four invasive alien squirrels (*Sciurus carolinensis* (grey squirrel), *Callosciurus finlaysonii* (Finlayson's squirrel), *Callosciurus erythraeus* (Pallas's squirrel) and *Tamias sibiricus* (Siberian chipmunk)) were reduced when the interactions between land-use and climate changes were included in models. In this case, climate-only models fail to account for lack of connectivity between habitats and limited overall habitat suitability, and may lead to an overestimate of the potential suitable habitat (Febbraro *et al.*, 2019). In Korea, the potential spread of an invasive alien insect *Thrips palmi* (melon thrips) in agricultural areas was shown to be influenced by rising temperatures in winter. Increased winter temperatures enabled a longer overwintering period, allowing the species to spread further across increasingly connected agricultural areas (Hong *et al.*, 2019).

Climate change and land-use changes may also interact by creating positive feedback loops, reinforcing biological invasions. For example, in the tropical dry forests of Bolivia experimental fires were shown to enhance the abundance of the invasive alien African grass *Megathyrsus maximus* (Guinea grass) in plots subjected to selective logging compared to unlogged areas, suggesting that increasing fire risk under climate change may interact with deforestation resulting from land-use change to promote the spread of this species (Veldman *et al.*, 2009). Similarly, in dryland areas in South Africa, degraded landscapes and road corridors invaded by alien grasses may alter the fuel characteristics sufficiently for fires to become a threat to an otherwise fire-absent vegetation type (Rahlao *et al.*, 2009, 2014). In these disturbed environments, land-use change and climate change interact to favour biological invasions.

3.5.2 Land-use change, climate change and nutrient pollution

While many studies comment upon the role of land-use change, climate change and nutrient pollution in the context of biological invasions, few explicitly capture interactions between these drivers or suggest causality for specific alien species introductions or stages of the biological invasion process. In fact, in many instances, invasive alien species are considered as a driver, rather than the response affected. In lieu of direct observations, paleoecological records may provide evidence for the interactive effects of land-use or sea-use change, nutrient pollution and climate change coinciding with an increase in invasive alien species. The analysis of a marsh sediment core spanning the last 1,100 years from Tivoli Bay in the Hudson River shows that climate shifts and other anthropogenic drivers (i.e., land-use change and nutrient input) occur simultaneously, with a fivefold expansion of invasive alien plant species such as *Typha angustifolia* (lesser bulrush), *Phragmites australis* (common reed) and *Lythrum salicaria* (purple loosestrife) (Sritrairat *et al.*, 2012). This study suggests that the increase in the number of invasive alien species is linked to European settlement impacts in this region, including higher disturbance, increased nutrients and sedimentation, along with warmer climate, however an explicit causal link between the different drivers and establishment of invasive alien species cannot be made.

A contemporary study of the seed recruitment of the invasive alien forbs *Bellis perennis* (common daisy), *Lolium perenne* (perennial ryegrass), *Poa pratensis* (smooth meadow-grass), *Taraxacum officinale* (dandelion), *Trifolium pratense* (red clover) and *Trifolium repens* (white clover) beyond their current invaded range in the subarctic Andes found that the species produce more biomass and flowers at higher nutrient levels and warmer temperatures and establish, grow and flower more in disturbed habitats than in undisturbed habitats (Lembrechts *et al.*, 2016). The study found no differences between the responses of these species when expanding in their invaded range in the Andes compared to expanding in their native range in the Scandes, suggesting that both plant invasions and natural range expansions in cold-climates are likely to increase with a combination of warmer climate, increased disturbance and increased nutrients (Lembrechts *et al.*, 2016).

In aquatic systems, the invasive alien *Phragmites australis* (common reed) is expanding throughout the Great Lakes of the United States (Great Lakes Phragmites Collaborative,

Box 3 12 **Land-use change, climate change and nutrient pollution interact to drive the introduction, establishment and spread of *Pontederia crassipes* across Africa.**

Pontederia crassipes (water hyacinth) is a fast-growing floating aquatic plant native to South America that has spread throughout vital freshwater bodies and wetlands of Africa, North America, Europe, Asia and Oceania since the late 1800s (Navarro & Phiri, 2000). Across Africa, this species has shown true exponential expansion by spreading over a large proportion of the water bodies with infestations getting worse as there is an increase in extreme climatic events in major water bodies like Lake Victoria in East Africa, Lake Nyasa in the Nile basin (especially around Lake Tana), the Zambezi River basin in southern Africa and the Tano lagoon and River Niger in West Africa. Connectivity among diverse water bodies has further facilitated the spread of water hyacinth in the region. Its spread is linked to eutrophication emanating from poor land-use management practices and is facilitated by environmental degradation and extreme climatic (i.e., temperatures, wind and floods) events (Navarro & Phiri, 2000; Téllez *et al.*, 2008; Thamaga & Dube, 2018). These extreme events facilitate the transport and introduction of water hyacinth in many freshwater ecosystems, and are also expected to alter natural surface water flow regimes, potentially further increasing the likelihood of water hyacinth and other invasive alien species establishing and spreading (Diez *et al.*, 2012; IPBES, 2019). Water hyacinth is expected to continue expanding into suitable habitat found in African ecosystems, with the rate and extent of the spread depending on disturbance from climate change and the nutrient pollution levels of the water bodies (Diez *et al.*, 2012; IPBES, 2019; Navarro & Phiri, 2000).

Figure 3 27 ***Pontederia crassipes* (water hyacinth) in Lake Victoria, Kisumu, Kenya.**

The free-floating invasive alien plant hinders small boats from docking and prevents fishing activities along the landing beaches. Photo credit: Mwe17, WM Commons – under license CC BY-SA 4.0.

2022), with its introduction, establishment and spread promoted by road networks, agricultural activities that increase nutrient availability and climate change (Mazur *et al.*, 2014). In Lake Victoria, Africa, the distribution of the invasive alien *Pontederia crassipes* (water hyacinth) rapidly expanded during the 1980s, a period linked to eutrophication and climate warming (Hecky *et al.*, 2010; Ogutu-Ohwayo *et al.*, 2016; A. E. Williams *et al.*, 2005; **Box 3.12** . While currently its spread is reduced, fears remain that land-use changes in the catchment area, along with continued nutrient loading and climate warming, will result in a resurgence of water hyacinth **(Box 3.12)**. In the late 1980s, the Baltic Sea experienced a bloom of *Mnemiopsis leidyi* (sea walnut) and a collapse of anchovy stocks, both of which were linked to a complex interaction of increased eutrophication, a changing regional climate with more severe winters and fishing pressures (Oguz *et al.*, 2008). Changing salinity arising from the new Suez Canal (**section 3.3.1.3**, **Box 3.7**) opening in conjunction with climate change is facilitated the expansion of *Brachidontes pharaonis* (variable mussel)'s distribution across the Mediterranean Basin, with this expansion also being enhanced by eutrophication associated with local urbanization (Sarà *et al.*, 2018).

3.5.3 Trade, urbanization and land-use change

Although international trade can directly introduce invasive alien species in ballast water, contaminants of commodities or stowaways in containers (**section 3.2.3.1**), it also interacts with other drivers facilitating biological invasions including: direct exploitation of natural resources, pollution, climate change, land-use change and urbanization **(Figure 3.28)**. To illustrate aspects of these interactions, the following section examines the implications of the interactions between trade, land-use change and urbanization.

International trade is an important driver of urbanization since it encourages the agglomeration of economic activities (and hence labour) in specific urban areas, particularly areas that are associated with international transport hubs such as marine ports, airports or national borders (Tripathi, 2020). Cities that have a high number of global trade links tend to be highly urbanized and urbanization also increases with the level of agricultural imports and with exports of non-agricultural commodities (Thia, 2016). Urban areas also represent hotspots of alien species richness, which in part can be explained by the high rate of intentional introductions of alien species

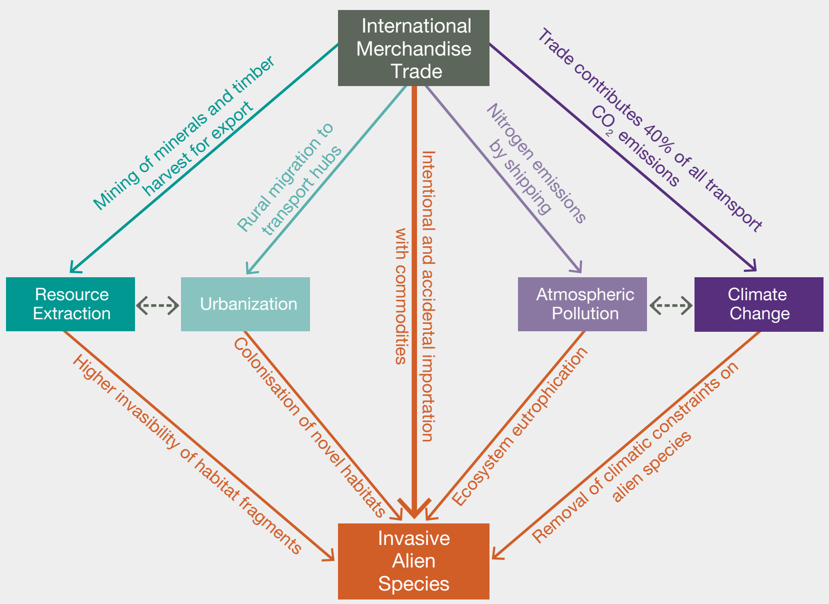


Figure 3 28 **Schematic illustration of how international merchandise trade interacts with other drivers of change in nature to influence the introduction and spread of invasive alien species.**

The thick arrow indicates the direct influence of international merchandise trade on invasive alien species, and thin arrows indicate how international merchandise trade affects other drivers which again may facilitate the introduction and spread of invasive alien species. Dotted lines represent interactions among drivers of change in nature. Adapted from Hulme (2021b), https://doi.org/10.1016/j.oneear.2021.04.015, under license CC BY 4.0.

either for amenity value (e.g., street trees) or as pets (e.g., parrots) that subsequently escape, but also by the higher international connectivity of large cities that facilitates unintentional introductions of alien species *via* ports and airports (Shochat *et al.*, 2010). In the marine realm, urbanized maritime infrastructure associated with ports (e.g., breakwaters, jetties and seawalls) does not function as a surrogate for natural rocky habitats but instead facilitates the establishment and spread of alien species (Bulleri & Chapman, 2010; **section 3.3.1.4**). For example, in coastal North America approximately 90 per cent of the alien species inhabiting hard artificial substrata have been reported from docks and marinas (Mineur *et al.*, 2012). Similarly, it can be expected that urbanization driven by international trade will lead to further development of land-based transport infrastructure such as rail and roads which may also facilitate the spread of alien species beyond the initial port of entry. The growth of urban areas also results in land-use change as natural and agricultural areas are fragmented and converted to housing. This environmental disturbance favours the persistence of generalist human-commensal species from around the world (Gavier-Pizarro *et al.*, 2010).

The extensive clearing of tropical forests in recent decades is in part driven by increased international trade in agricultural commodities and this trend is expected to continue due to further trade liberalization (Schmitz *et al.*, 2015). Increasing global demands for meat, animal feed and oil seed products have led to major changes in land-use in developing countries (Pendrill *et al.*, 2019). There is also a link between international trade in wood products (particularly roundwood timber) and declining national forest stocks, especially in developing countries in the tropics such as Indonesia and Cameroon (Kastner *et al.*, 2011). In addition, international trade increases demand for new products as in the case of the expansion of oil palm plantations in Latin America, which has occurred at the expense of other land-uses including tropical forests (Furumo & Aide, 2017). The resulting fragmentation of tropical forests also increases their vulnerability to biological invasions (Waddell *et al.*, 2020). Furthermore, new crops such as *Elaeis guineensis* (African oil palm) can themselves spread beyond cultivated areas to become invasive alien species in regions where they are not native (Zenni & Ziller, 2011).

3.5.4 Urbanization and pollution

Urbanization and pollution interact to facilitate biological invasions; invasive alien species are disproportionally found in urbanized areas with higher pollution compared to less polluted urban areas or polluted natural ecosystems. While urbanization promotes the transport and introduction and of invasive alien species both intentionally and unintentionally (**section 3.2.2.4**), pollution contributes to improve the chances of establishment and spread of an invasive alien species, which tend to be facilitated by nutrient-rich habitats (**section 3.3.3.1**). For example, in reef ecosystems, alien faunal distributions are linked to the presence of heavy metals, local population density and proximity to city ports; with invasive alien species being more common in areas with higher levels of pollution, while native species are less common under these conditions (Stuart-Smith *et al.*, 2015). In mangrove ecosystems in Nigeria, pollution and urbanization create forest gaps that enhance biological invasions, for example of *Nypa fruticans* (nipa palm; Nwobi *et al.*, 2020).

Urban areas generate and disseminate many types of pollutants, including nitrogen. Alien species are generally more tolerant to nitrogen pollution than native species (**section 3.3.3.1**), and in nitrogen polluted freshwater ecosystems in Hawaii, native goby species have declined while alien species are increasing (Lisi *et al.*, 2018). Urbanization generates runoff to aquatic ecosystems, which can carry many pollutants, and modifies their dynamics. An example is how pollution of freshwater environments following the application of road de-icing salts facilitates the survival and establishment of the invasive alien *Corbicula fluminea* (Asian clam) in New York, the United States, as this species is more tolerant to road salts than other native freshwater organisms (Coldsnow & Relyea, 2018).

Pollution can act directly by conferring advantages to more tolerant invasive alien species, but also by creating a competitive advantage for them by negatively affecting native populations. Human settlements generate domestic wastewater, agricultural fertilizer runoff, and effluents enriched with organic nutrients that, in coastal urban areas, often end up directly in the sea. In South Africa, these inputs of organic pollutants are associated with the bloom of *Ulva lactuca* (green laver) in Saldanha Bay (Mead *et al.*, 2013, **section 3.3.3.1**). In the Mediterranean Sea, high levels of urbanization are also linked with the degradation of coralligenous assemblages, compared to sites within natural protected areas and areas with lower rates of urbanization. This difference is, at least partially, associated with the increase of opportunistic alien species (e.g., algal species), which are more tolerant to urban-related pollution (Montefalcone *et al.*, 2017). In South East Australia, the heavily urbanized Port Jackson Estuary is one of the world's waterways most polluted by heavy metals and organic compounds as a result of antifouling paint, and this threatens the native *Saccostrea glomerata* (Sydney rock oyster), giving the more tolerant invasive alien *Magallana gigas* (Pacific oyster) a competitive advantage, and the invasive oyster, which also takes advantage of artificial substrates to establish has become more abundant (Scanes *et al.*, 2016). In coastal urbanized areas, copper, which is one of the primary active ingredients in antifouling hull paints, has become a common pollutant that increases in concentration as vessel traffic increases. A study carried out in Massachusetts proved that the invasive alien *Botrylloides violaceus* (violet tunicate) poses a competitive threat to the native *Aplidium glabrum* (tunicate) with regard to surface area growth when copper pollution is present. *Botrylloides violaceus* proved to be more tolerant to copper pollution (Osborne & Poynton, 2019). In Tasmania, expanding urbanization close to estuaries has resulted in an increase in pollutants from anthropogenic sources (e.g., marinas, storm-water drains, sewage outfalls and fish farms) that affect nearby benthic assemblages. Alien species were more abundant in sites near marinas and sewage outfalls (Fowles *et al.*, 2018). Increased focus on controlling pollution from marinas and sewage outfalls may thus limit the spread of alien species (Fowles *et al.*, 2018). In Lane Cove Valley, Australia, invasive alien plants are linked to areas polluted by phosphorus and heavy metals, which are pollutants linked with urbanization, and especially to areas where soil has also been previously disturbed (S. J. Riley & Banks, 1996).

Plastics are also a common waste product in urban areas, and exposure to plastics can change the behaviours of species. In Chile, Pinochet *et al.* (2020) studied invasive alien bryozoan species, such as *Bugulina flabellata* and *Bugula neritina* (brown bryozoan), that are frequently found in urbanized areas globally. These species tend to prefer plastic substrates, rather than wood or concrete, and exposure to plastic substrates could enhance their spread (**section 3.3.3.4**).

Finally, there are other types of pollution less frequently studied that are also linked to urbanization such as noise and light pollution. There are a few studies that assess the link between noise and light pollution in urban areas and invasive alien species. For example, the invasive alien *Hemidactylus frenatus* (common house gecko) in north-eastern Australia occupies a broader range of light environments than does the native gecko *Gehyra dubia* (dubious dtella). Experimental removal of the invasive alien gecko from places with light pollution, did not change the selection by native geckos for darker locations, which suggests cities are opening niches for invasive alien species (Zozaya *et al.*, 2015).

3.6 SYNTHESIS AND CONCLUSION

3.6.1 Literature used in this chapter and identification of knowledge gaps

A diverse strategy was adopted to identify and summarize the literature used in Chapter 3 (**sections 3.1.3** and **3.1.4**).[9] The varying approaches to reviewing the literature not only recognized the biases and gaps within the scientific literature linking invasive alien species with other drivers of change in nature (Hulme, 2022; **Box 3.13**), but also acknowledged the complexities in establishing cause-effect relationships between drivers and the transport, introduction, establishment and spread of invasive alien species (**sections 3.1.2** and **3.1.4**, **3.1.5**). Based on these search strategies, Chapter 3 summarizes the information from 1,183 scientific papers and other sources9 across indirect, direct, and other drivers of change in nature for the role of a total of 44 drivers in facilitating the transport, introduction, establishment and spread of invasive alien species across biomes, realms and IPBES regions (**sections 3.2** and **3.3**, **3.4**, **3.5**). Sections 3.2, 3.3, 3.4, 3.5 and 3.6.2 appraise and summarize the current state of knowledge on the role of drivers in facilitating biological invasions, whereas sections 3.1 and 3.6.1 outline the background and search strategies and summarize the knowledge base for the chapter.

Across the 1,183 studies identified and used in this report **(Figure 3.29)**:

- 30.4 per cent reported on indirect drivers of change in nature (sociocultural 2.2 per cent, demographic 9.5 per cent, economic 10.4 per cent, technology 5.9 per cent, governance 2.4 per cent);
- 72.1 per cent on linked to direct drivers of change in nature (land- and sea-use change 29.3 per cent, direct exploitation of natural resources 7 per cent, pollution 12.1 per cent, climate change 17.2 per cent, invasive alien species 6.5 per cent);
- 5.8 per cent on linked to other drivers (biodiversity loss 3.9 per cent, natural drivers 1.9 per cent).

Despite targeted searches to address how two or more drivers of change in nature interact to facilitate biological invasions (**sections 3.1.5**, **3.5**) fewer than 20 per cent of these studies reported on the role of more than one driver in facilitating biological invasions (**Boxes 3.12** and **3.13**).

9. Data management report available at: https://doi.org/10.5281/zenodo.5529309

Most studies were on invasive alien plants (52.8 per cent), which represents almost twice as many as for invasive alien vertebrates (28.9 per cent) or invertebrates (29.2 per cent), with fewer studies having been conducted on invasive alien microbes and fungi (12.3 per cent). More than half of the identified studies were from the terrestrial realm (66.3 per cent), whereas freshwater and marine systems were represented by, respectively, 20.2 per cent and 23.7 per cent of the studies. Plants dominated studies of terrestrial invasive alien species, whereas all macroscopic taxonomic groups were relatively evenly represented in the studies reviewed from the aquatic realms, with vertebrate studies being most numerous in the freshwater realm and invertebrate studies most numerous in the marine realm **(Figure 3.30A)**.

Reviewed studies mostly focused on drivers in the Americas (45 per cent), followed by Europe and Central Asia (36.6 per cent), with fewer studies from the Asia-Pacific region (34.5 per cent, noting that 23.2 per cent were from Oceania) and Africa (26 per cent). This trend was relatively consistent across all taxonomic groups **(Figure 3.30B)**.

For the majority of the studies analyzed in this report (1044 sources), it was possible to link drivers to one or more stages of the biological invasion process, with the majority of studies linking drivers to the establishment (70.5 per cent), spread (66.4 per cent) and introduction (53.5 per cent) of invasive alien species, whereas fewer sources explicitly linked drivers to transport (33.8 per cent). These numbers add up to considerably more than 100 percent, illustrating that most studies link drivers to more than one stage within the biological invasion process. The availability of studies on drivers varied across stages of the biological invasion process. Studies of the role of indirect drivers of change in nature in facilitating biological invasions tended to focus on links to transport and introduction of invasive alien species **(Figure 3.31A)**. This trend was especially evident for sociocultural and economic drivers, for which studies on their role in transport and introduction made up for two thirds (65 per cent) of studies on these two drivers across all stages of the biological invasion process. This pattern was consistent across all realms and taxonomic groups but is especially strong for microbes. In contrast, studies of direct (anthropogenic and other) drivers, reported

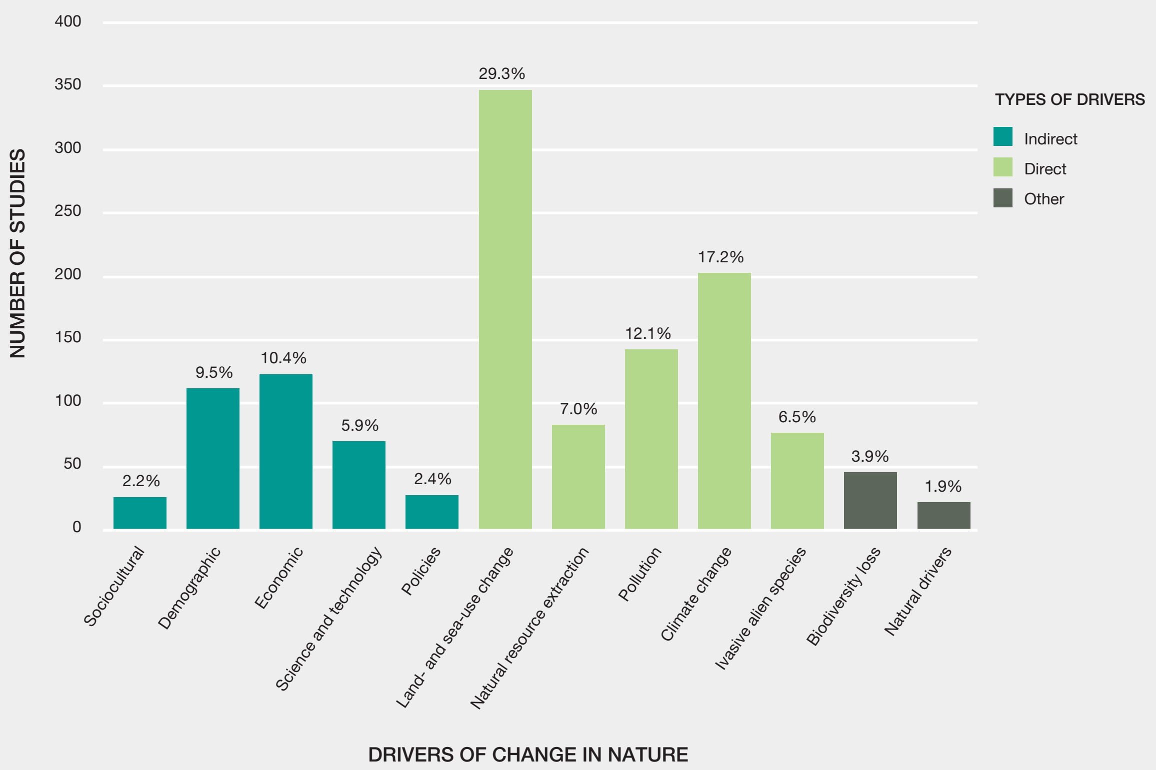


Figure 3 29 **Overview of the literature used for this assessment of the drivers of change in nature affecting biological invasions.**

The number of studies (y axis) in the literature base for Chapter 3 reporting on each driver of change in nature (x axis) in affecting biological invasions. The total number of studies is 1185. The underlying data for this figure is available at https://doi.org/10.5281/zenodo.5529309

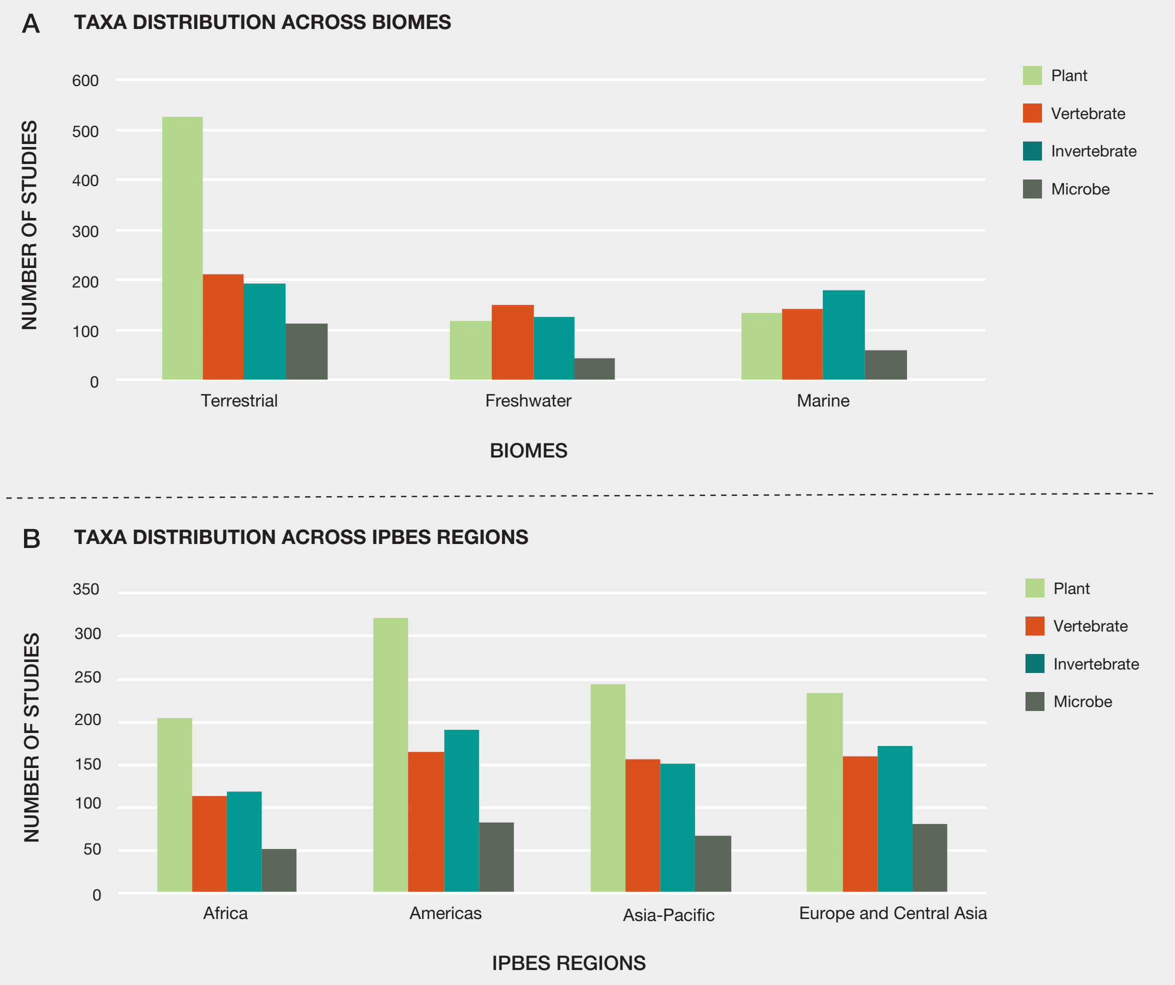


Figure 3 30 **Overview over the literature used for this assessment of the role of drivers of change in nature in affecting biological invasions across realms, IPBES regions and taxa.**

The number of studies in the literature base for Chapter 3 (y axis) reporting on the role of drivers of change in nature in facilitating invasive alien species categorized by realms vs. taxonomic groups (panel A.; x axis) and by IPBES regions vs. taxonomic groups (panel B.; x axis) (n = 1202 papers).The underlying data for this figure is available at https://doi.org/10.5281/zenodo.5529309

predominantly on links to the establishment and spread stages, a pattern that was especially evident for pollution, climate change, invasive alien species, biodiversity loss and ecosystem resilience, and for natural drivers, for all of which studies on their role in establishment and spread made up more than 75.9 per cent of all studies (**Figure 3.31B** and **C**). These patterns were less consistent across realms and taxonomic groups, however, as stronger links between direct or other drivers and establishment and spread were found in the terrestrial realm than in freshwater, whereas this pattern was largely absent in marine systems. Likewise, the link between indirect drivers and the early stages of biological invasion and direct drivers and later stages was strong for alien plants, but less evident for invertebrates and vertebrates, whereas studies of microbes predominantly reported on links to transport and introduction across all drivers. Three groups of drivers, demographic drivers, land- and sea-use change and direct exploitation of natural resources, stood out as studied in relation to all stages of the biological invasion process (**Figure 3.31A** and **B**). This was less evident for land-use and direct exploitation of natural resources in the terrestrial realm and for plants, where studies of establishment and spread dominate, and for demographic drivers for microbes, where studies of transport and introductions, dominated.

Information on the roles of drivers in facilitating biological invasions was largely sourced from primary studies (50.4 per cent), followed by reviews (40.9 per cent), whereas only a small number of studies were formal meta-analyses (3.6 per

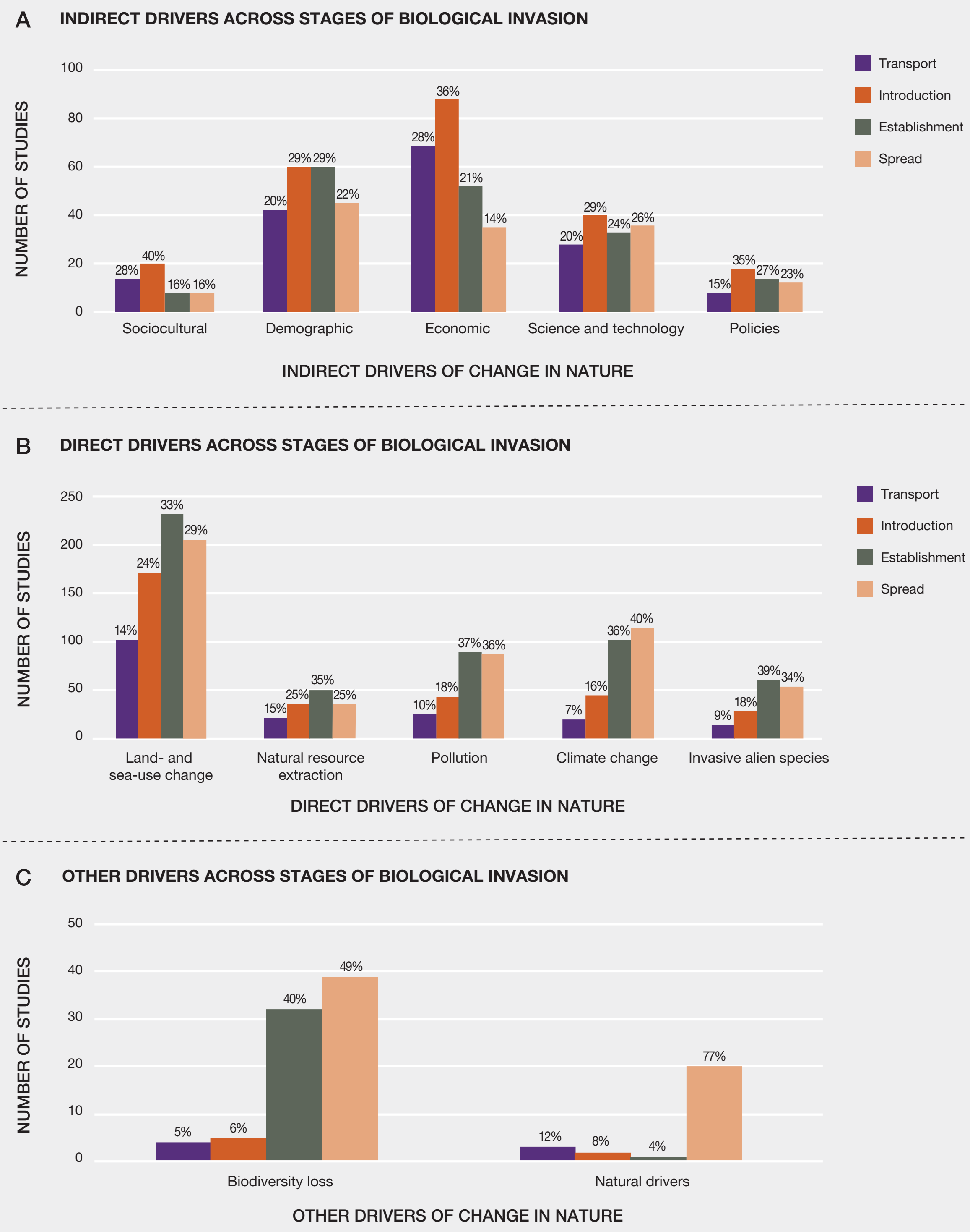


Figure 3 31 **Overview of the literature used within this chapter and specifically the role of drivers of change in nature in affecting invasive alien species across the biological invasion process.**

Summary of the literature extracted for Chapter 3 for assessing the role of drivers across biological invasion stages, showing the proportion of the studies of each driver that addresses the different stages in the biological invasion process. Note that studies used in the chapter that do not explicitly state stages of the biological invasion process affected are excluded from this figure, and that each study can address more than one stage of the biological invasion process (total of 1,183 papers). Chi-square analysis indicated

Figure 3 31

that the number of studies reporting effects on the transport and introduction of invasive alien species is statistically higher for indirect than direct drivers of change in nature ($p<0{,}001$), whereas the opposite is true for the establishment and spread stages. The number of studies is shown on the y axis, while the indirect (panel A.), direct (panel B.) and other drivers (panel C.) of change are shown on the x axis. For the number of studies for each driver, see **Figure 3.29**. The underlying data for this figure is available at: https://doi.org/10.5281/zenodo.5529309

cent). This suggests further structured synthesis and reviews are required to support the knowledge on the drivers facilitating biological invasions **(Box 3.13)**.

The outcomes of the chapter review with respect to data gaps and biases in the evidence available for assessing the role of drivers of change in nature in the context of biological invasions, are largely consistent with both the systematic review of scenarios and models[10] (**Chapter 1**, **section 1.6.7.3**) undertaken as part of this assessment **(Box 3.14)** and the bibliometric analysis of research effort undertaken by Hulme (2022) on drivers in relation to biological invasions **(Box 3.13)**.

10. Data management report available at https://doi.org/10.5281/zenodo.5706520

Box 3 13 **Impacts of direct and indirect drivers of change in nature on biological invasions are currently much less understood than other areas of conservation science.**

A recent assessment of the research effort into the role of indirect and direct drivers of change in nature facilitating invasive alien species concluded that the current knowledge is limited, and focuses on tractable drivers over those that require an interdisciplinary approach, with bias towards developed economies (Hulme, 2022). Between 2000 and 2020, 27,462 peer-reviewed journal articles were published addressing biological invasions of which less than 5,000 (or 18 per cent) examined the role of one or more drivers of change in nature. In contrast, out of a corpus of 110,087 research papers targeting biodiversity and ecosystem services, almost 40,000 (or 36 per cent) described the role of one or more drivers. Thus, the drivers affecting biological invasions remain less understood compared to other areas of conservation science.

Research on drivers of change in nature facilitating biological invasions reflects a strong bias towards direct drivers with only a small fraction of studies addressing indirect drivers (Hulme, 2022). While there have been calls for increased interdisciplinarity in the study of biological invasions, the percentage of articles addressing indirect drivers of change in nature has shown no significant increase over the last two decades, leading to an increasing bias in articles towards direct drivers of change in nature (Hulme, 2022). Drivers deemed likely to be important for biological invasion by invasive alien species, such as governance and direct exploitation of natural resources, were shown to be poorly supported by research effort. The considerable literature addressing national and international policies for conserving biodiversity (Le Preste, 2017) is not matched by similar studies tackling the governance of problems arising from invasive alien species (Hulme, 2021a).

Compared to developed economies, there were only about half as many articles affiliated with institutions in developing economies. This may significantly limit the opportunity for prevention and projection of future risk of invasive alien species in Africa, Asia and Latin America. Given the future importance of indirect drivers such as tourism, trade and infrastructure projects on the likely risk of introducing invasive alien species to developing economies (Hulme, 2015a), the paucity of studies on indirect drivers is particularly troubling. Developing economies harbour most of the world's biodiversity (Adenle *et al.*, 2015) but also face significant threats from indirect drivers of change in nature such as international trade (Lenzen *et al.*, 2012) as well as direct drivers in the form of invasive alien species (Early *et al.*, 2016). Developing economies have also been identified as sources of many of the world's invasive alien species that have the potential to reach nearly all terrestrial biomes (Measey *et al.*, 2019). Thus, there is an imperative to improve the knowledge of drivers of change in nature in developing economies not only to protect their own national natural heritage but also prevent further biological invasions globally. The similarity between the results from the diverse literature review strategy in Chapter 3 and a systematic bibliometric analysis (Hulme, 2022) support the view that Chapter 3 adequately captures current knowledge of the drivers in facilitating invasive alien species.

Box 3 14 Representation of drivers in scenarios and models.[11]

Previous IPBES assessments have evaluated how various tools and techniques such as scenarios and models have been used to better understand the impacts of drivers on nature, nature's contributions to people and good quality of life (IPBES, 2016b, 2018e, 2018c, 2018d, 2019). For this assessment a systematic review was undertaken to evaluate the patterns and trends in published research on invasive alien species that included both scenarios and models (**Chapter 1**, **section 1.6.7.3**). In total 778 papers were reviewed (from an initial set of 30,299). Drivers were included as a scenario feature within the review with information captured on the number and type of driver. The results of the review showed that most papers focused on only one driver (77 per cent of 778 papers). Climate change was the most commonly included driver (62 per cent of all observations; **Figure 3.32**). The studies that focused on two or more drivers often included climate change or invasive alien species. Many of the drivers identified in **Chapter 3** **(Table 3.1)** were poorly represented, as drivers such as demographics, governance, pollution, direct exploitation of natural resources, values and technology accounted for less than 2 per cent of the observations. The lack of studies focusing on interactions amongst drivers is a gap that could limit understanding of the outcomes of biological invasions alongside other drivers of change in nature.

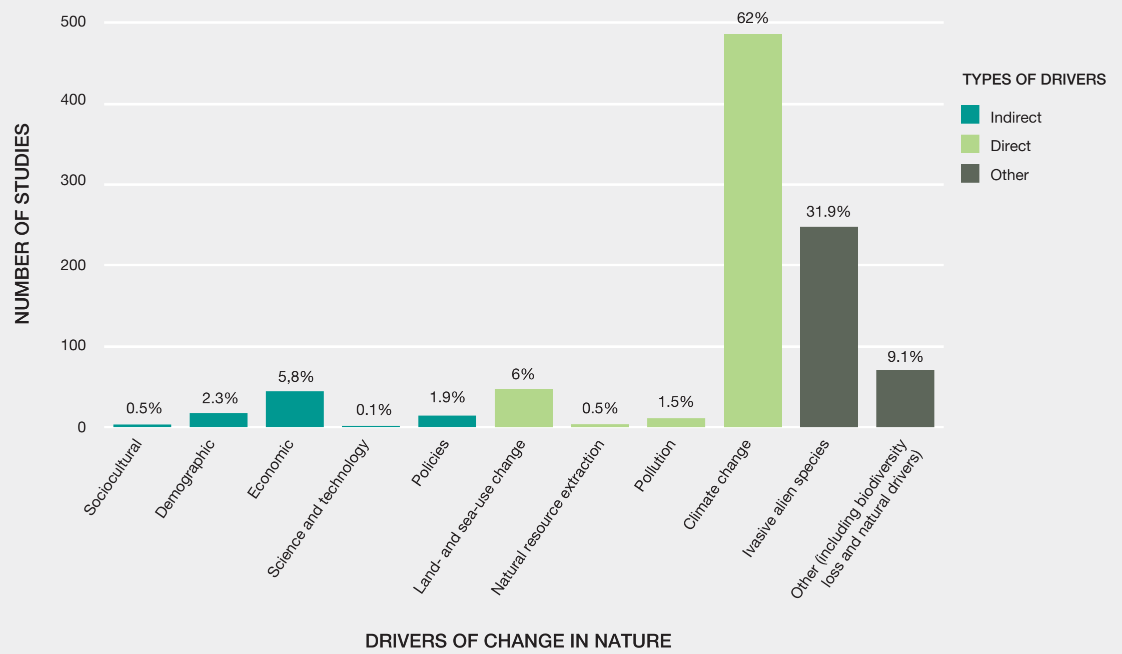


Figure 3 32 **Representation of drivers (per cent) of the observations for each driver across the papers included in the scenarios and models review noting some papers included multiple observations.**

The number of studies is shown on the y axis, and the different drivers of change in nature are shown on the x axis. A data management report for this figure is available at https://doi.org/10.5281/zenodo.5706520

The dominance of climate change as a driver in most studies is explained by the prevalence of correlative models which invariably include climate change scenarios (e.g., scenarios from the Intergovernmental Panel on Climate Change; IPCC) as a factor to project the occurrence or potential distribution of species. The majority of papers focused on exploratory scenarios, that examine a range of plausible futures based on the potential trajectory of key underlying scenario features. The results of these studies provide some insights in relation to future directions for the development of scenarios and models in invasion science (e.g., Lenzner *et al.*, 2019; Roura-Pascual *et al.*, 2021). Future studies may be able to address the current gaps and include the other cross-cutting themes highlighted in this assessment such as Indigenous and local knowledge (**Glossary**) and good quality of life (e.g., Obermeister, 2019), which would improve our understanding of the patterns and trends in drivers of change in nature and how these affect biological invasions.

11. Data management report available at https://doi.org/10.5281/zenodo.5706520

3.6.2 Synthesis

The relative importance of drivers of change in nature in facilitating biological invasions was quantified based on a consensus approach involving an expert-based assessment by the authors of the chapter for each invasion stage across and within realms (terrestrial, freshwater, marine) and broad taxonomic units (microbes, plants, invertebrates, vertebrates).[12]

Overall, the expert-based consensus approach assessed economic drivers as the most important in facilitating biological invasions worldwide (average 21 per cent importance, across all realms and taxonomic units, and stages of the invasion process), followed by land- and sea-use change (16 per cent), demographic drivers (10 per cent), climate change (9 per cent), sociocultural drivers and policies, governance and institutions (each 8 per cent), pollution (7 per cent), direct exploitation of natural resources and invasive alien species (each 5 per cent), biodiversity loss, natural hazards and science and technology (each 4 per cent). The major drivers identified are consistent with indigenous and local knowledge **(Box 3.15)**.

The consensus approach further reveals a clear shift in relative importance of the drivers over the stages of the biological invasion process **(Figure 3.34)**. The transport and introduction of invasive alien species are primarily facilitated by economic drivers, followed by land- and sea-use changes, with some evidence for an additional role of sociocultural drivers, demographic drivers, and policies, governance and institutions. In contrast, land- and sea-use change is the overriding driver responsible for the establishment and spread of invasive alien species, followed by climate change, pollution, and to some extent economic drivers and biodiversity loss. Thus, indirect drivers are identified as the most important in the early stages of the biological invasion process while direct drivers dominate in the later stages.

Patterns are relatively consistent across realms, but with some variation **(Figure 3.34A)**. Economic drivers and natural drivers are considered relatively more important and sociocultural and policy, governance and institutional drivers less important for facilitating the transport of invasive alien species in the marine realm, whereas demographic drivers are more important for the transport of invasive alien species into the terrestrial realm. There is little difference between realms in the relative importance of drivers facilitating the introduction stage. Policies, governance and institutional drivers are more important in facilitating the establishment of invasive alien species, while sociocultural drivers and biodiversity loss are less important in the marine realm, whereas demographic drivers are relatively more important and direct exploitation of natural resources less important in the terrestrial realm. Finally, the spread of invasive alien species is relatively less affected by demographic drivers in the marine realm, more affected by sociocultural drivers in freshwater systems, and less affected by pollution but more affected by biodiversity loss in the terrestrial realm.

There is more variation across taxonomic units than realms **(Figure 3.34B)**. Sociocultural drivers are consistently more important for facilitating alien plants early in the invasion process and for vertebrates across all stages, and less important for microbes and invertebrates. This pattern likely reflects the importance of intentional introductions of plants and animals for human amenity values, both linked to subsistence and to cultural values (**sections 3.2.1**, **3.2.3.2**, **3.2.3.3**, **3.3.1.1**). Demographic drivers are relatively more important for the transport of microbes than other taxonomic groups, whereas economic drivers are relatively less important for the introduction of plants and more important for the spread of plants and vertebrates than for the other taxonomic groups. This latter finding is likely linked to the relatively higher importance of land-use changes for the introduction of plants along with intentional introductions of alien vertebrates for hunting and farming (**sections 3.3.1.1**, **3.3.2.1.1**, **3.3.2.1.2**). Pollution is assessed as relatively more important for the introduction of microbes and less important for the establishment of invasive alien vertebrates and the spread of invasive alien invertebrates and vertebrates. In contrast, climate change is deemed relatively more important for the introduction, establishment and spread of microbes compared to other taxonomic groups.

These patterns illustrate some of the complexity in how drivers of change in nature facilitate biological invasions. The variation across stages, taxonomic units and realms is partly related to the biophysical characteristics of the specific processes and systems. As examples, trade and travel operate mainly through both the intentional and unintentional transport of invasive alien species across regions; harvesting and restocking for harvesting are more important in aquatic than terrestrial systems (**sections 3.2.3.1**, **3.3.2.1**) and respond to variation in human impacts (such as, land- and sea-use being more important for plants and pollution within aquatic systems and less important for vertebrates; **sections 3.3.1.1**, **3.3.3**).

12. Data management report available at https://doi.org/10.5281/zenodo.7861162

Box 3 15 Identification of drivers by Indigenous Peoples and local communities.[13]

The IPBES framework acknowledges diverse knowledge sources in assessments, and in particular the central position of Indigenous Peoples and local communities in providing situated understanding of biological invasions. Assessment authors thus carried out an extensive cross-chapter review of literature to identify Indigenous and local knowledge related to invasive alien species (**Chapter 1**, **section 1.6.7.1**). In total 131 studies were reviewed, and data on drivers was collated, including both the number and type of driver identified along with any comments or additional information.

In most cases, Indigenous Peoples and local communities identified at least one driver that had facilitated the invasion of the reported alien species (84 per cent of 131 papers). Land-use change was the most commonly included driver (identified in 40 per cent of all papers), but Indigenous Peoples and local communities generally judge indirect drivers of change in nature as relatively important in facilitating biological invasions, with economic drivers (32 per cent), policies, governance and institutions (24 per cent) and sociocultural drivers (21 per cent) as the three next-ranked drivers **(Figure 3.33)**. In the majority of studies (68 per cent), Indigenous Peoples and local communities identified more than one driver facilitating biological invasions, the average number of drivers being 2.25 (range 1-7). Of the studies, 72 per cent reported on the spread of invasive alien species, 18 per cent on their establishment, and 11 per cent on their introduction, with no studies reporting on the transport stage of the invasion process.

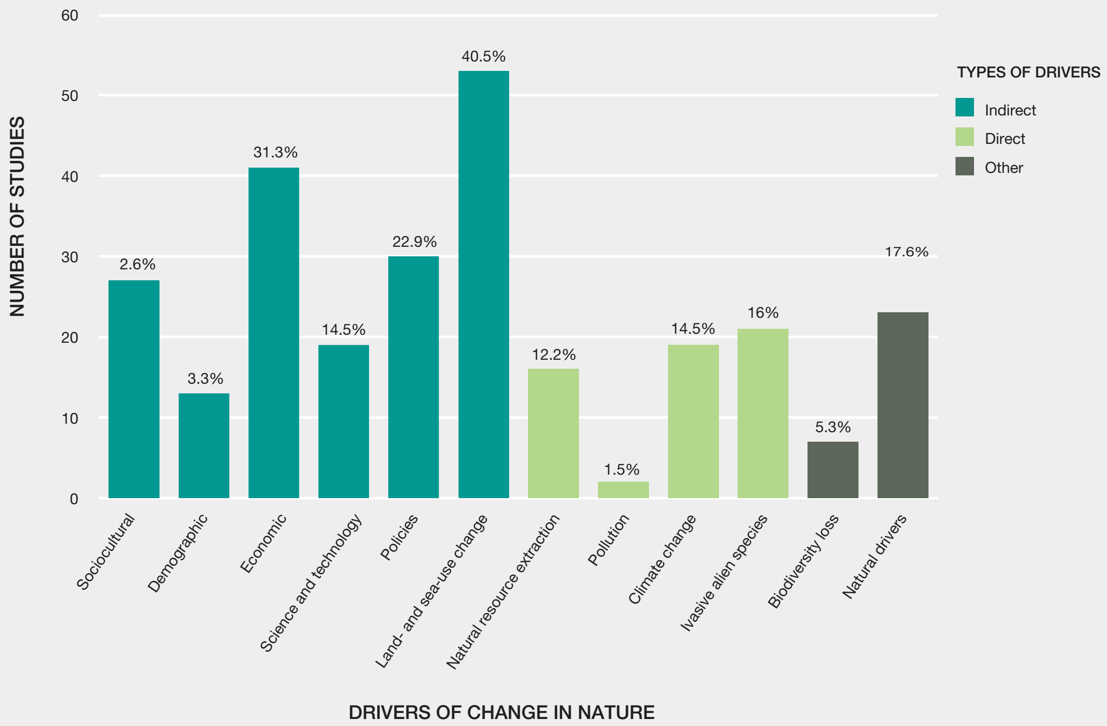
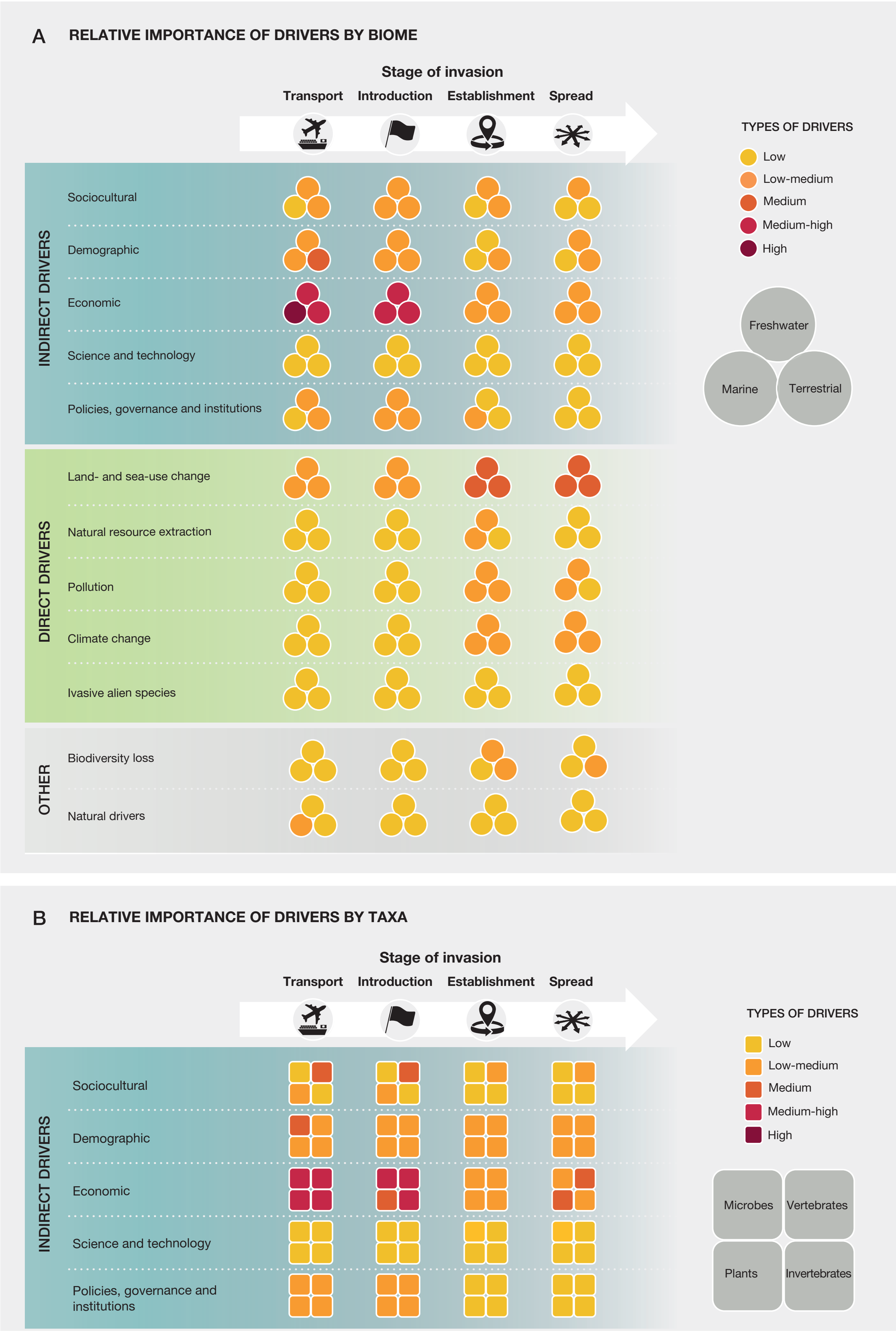


Figure 3 33 Representation of the role of each driver in the studies included in the literature review on Indigenous Peoples and local communities and invasive alien species (numbers of reported cases across 131 references) in facilitating biological invasions, noting most papers included multiple cases.

The number of studies is shown on the y axis, and the different drivers of change in nature are shown on the x axis. Data management report available at https://doi.org/10.5281/zenodo.5760266

13. Data management report available at https://doi.org/10.5281/zenodo.5760266


A
RELATIVE IMPORTANCE OF DRIVERS BY BIOME
Stage of invasion
Transport
Introduction
Establishment
Spread
INDIRECT DRIVERS
Sociocultural
Demographic
Economic
Science and technology
Policies, governance and institutions
DIRECT DRIVERS
Land- and sea-use change
Natural resource extraction
Pollution
Climate change
Ivasive alien species
OTHER
Biodiversity loss
Natural drivers
TYPES OF DRIVERS
Low
Low-medium
Medium
Medium-high
High
Freshwater
Marine
Terrestrial
B
RELATIVE IMPORTANCE OF DRIVERS BY TAXA
Stage of invasion
Transport
Introduction
Establishment
Spread
INDIRECT DRIVERS
Sociocultural
Demographic
Economic
Science and technology
Policies, governance and institutions
TYPES OF DRIVERS
Low
Low-medium
Medium
Medium-high
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Microbes
Vertebrates
Plants
Invertebrates

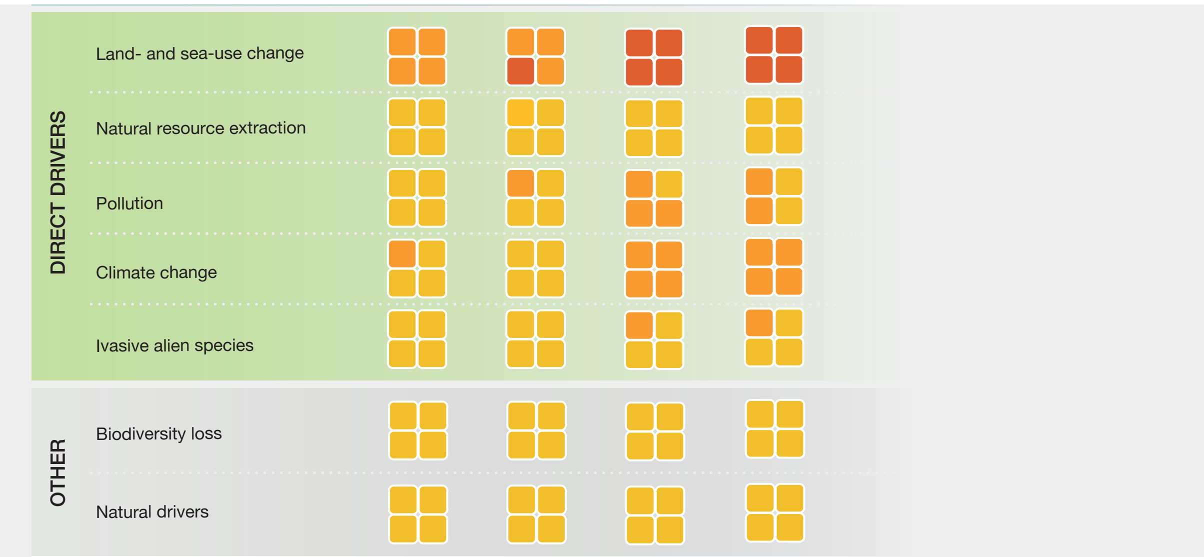

Figure 3 34 **Assessment of the relative importance of indirect and direct drivers of change in nature in facilitating invasive alien species across stages of the biological invasion process, by A) biome (terrestrial, freshwater, marine) and B) taxa (microbes, plants, invertebrates, vertebrates).**

Separately for each realm and taxonomic unit, the Chapter 3 experts (coordinating lead and lead authors, fellows), scored the relative importance of each driver (out of 100 points) for each of the four stages of the invasion process. The scores of all panel members were then averaged to produce an overall consensus assessment, based on which the authors were allowed to adjust their individual scores (i.e., using a Delphi method approach). A data management report for this figure is available at https://doi.org/10.5281/zenodo.7861162

3.6.3 Conclusions

Indirect and direct drivers of change in nature play significant but varying roles across all four stages in the biological invasion process **(Figure 3.34)**. There is a clear shift from the early stages (transport and introduction), where indirect drivers of change in nature play overriding roles, and the later stages (establishment and spread), where direct drivers dominate. The transport and introduction of invasive alien species are primarily facilitated by economic drivers (**section 3.2.3**), in particular, international trade, primarily though maritime commerce, which has caused both intentional and unintentional transport and introduction of many invasive alien species in both terrestrial and aquatic realms, followed by land- and sea-use change (**section 3.3.1**), with other drivers playing smaller but still significant roles **(Figure 3.34)**. Land- and sea-use change is the overriding driver for the establishment and especially spread of invasive alien species, followed by climate change, pollution and to some extent demographic and economic drivers **(Figure 3.34)**.

There is variation in relative importance of drivers across realms and taxonomic groups, partly related to the biophysical characteristics of the specific systems (such as better connectivity in aquatic than terrestrial systems) and partially related to variation in human impacts. Sociocultural drivers are consistently more important for plants and vertebrates than for microbes and invertebrates across all stages of the invasion process **(Figure 3.34)**. This pattern probably reflects the importance of intentional introductions for human amenity value. Recent research on the role of drivers of change in nature in facilitating biological invasions has focused on the direct drivers, climate change and land-use change, whereas economic drivers are the most studied indirect driver. The importance of governance and sociocultural perspectives in shaping biological invasion remains understudied. Strong biases also occur in the biomes and taxonomic groups examined, with the majority of studies dealing with terrestrial temperate ecosystems relative to other biomes, and plants relative to other organismal groups (**section 3.6.1**). The evidence base for the role of direct and indirect drivers on invasive alien species is largely drawn from developed nations, particularly Europe, the United States and Canada as well as Australia and New Zealand. While most indirect and direct drivers of change in nature affect biological invasions across all regions and ecosystems, the magnitude of their effects will differ and the lack of detailed information for the Arctic and developing nations, especially sub-Saharan Africa, tropical Asia and South America, is of concern.

Intensification of drivers and the acceleration of biodiversity loss and ecosystem degradation will have consequences for biological invasions in the future. It is becoming clear that climate change will increasingly shape future trends in invasive alien species, potentially with a significant temporal lag (**Chapter 2**, **section 2.2.1**), and will modify the role that other direct and indirect drivers might play in facilitating biological invasions. Furthermore, ecosystems may become more vulnerable to biological invasions as invasive alien species themselves decrease biotic resistance to further biological invasions and/or biodiversity is lost (**sections 3.3.5**, **3.4.2**). All these concurrent changes in drivers and ecosystems are indicative that past patterns of biological invasions may not be effective in informing future invasion patterns. Of particular concern is the lack of understanding as to how different drivers of change in nature interact to affect biological invasions across the invasion stages.

Few drivers act in isolation (**sections 3.1.5**, **3.5**), and there are potentially many interactions among drivers that are likely to lead to future biological invasions scenarios never previously experienced. For example, international trade also influences other drivers of change in nature that facilitate biological invasions by intensifying urbanization around major trade ports, driving resource extraction to meet international market demands and increasing atmospheric and aquatic pollution (**section 3.5**). Similarly, land- and sea-use change has led to changes in disturbance regimes and habitat degradation, which can decrease biotic resistance to the establishment and spread of invasive alien species (**section 3.3.1**). Increasing and expanding trade, travel and urbanization are major drivers implicated in the introduction and spread of invasive alien species worldwide that at the same time facilitate ecosystem degradation, which in itself is a direct driver facilitating biological invasions (**section 3.1.2**). However, fewer than 5 per cent of published studies examining the role of drivers in facilitating biological invasion addressed more than one driver (**section 3.6.1**).

Intensification in many co-occurring drivers of change in nature in combination with interactive effects amongst drivers increase the risk of positive feedbacks exacerbating biological invasions in the future. Addressing these complexities can be achieved through interdisciplinary collaborations including scientists and policymakers. For example, future scenarios can be used to explore how economic, policy and demographic changes alongside and in response to climate change, land- and sea-use change, or pollution might lead to greater risk of biological invasions (i.e., the patterns described in **Chapter 2**). Such scenarios could then enable identification of the specific conditions, situations and combinations of drivers that are key in facilitating biological invasions, and that would therefore be critical to address in order to reduce threats and impacts from invasive alien species (**Chapter 4**). Only by investing in building these links between science and policy can risks of unintended policy outcomes, that lead to environmental degradation and biodiversity loss, be identified and avoided (**Chapters 5**, **6**). Better orientation and coordination of national and international research on drivers in relation to both their actual importance as well as their policy relevance in relation to biological invasions by invasive alien species is therefore key to addressing biological invasions in the future.

REFERENCES

Abatzoglou, J. T., & Kolden, C. A. (2011). Climate Change in Western US Deserts: Potential for Increased Wildfire and Invasive Annual Grasses. *Rangeland Ecology & Management, 64*(5), 471–478. https://doi.org/10.2111/REM-D-09-00151.1

Abatzoglou, J. T., & Williams, A. P. (2016). Impact of anthropogenic climate change on wildfire across western US forests. *Proceedings of the National Academy of Sciences, 113*(42), 11770–11775. https://doi.org/10.1073/pnas.1607171113

Abellan, P., Carrete, M., Anadon, J. D., Cardador, L., & Tella, J. L. (2016). Non-random patterns and temporal trends (1912-2012) in the transport, introduction and establishment of exotic birds in Spain and Portugal. *Diversity and Distributions, 22*(3), 263–273. https://doi.org/10.1111/ddi.12403

Achury, R., Holway, D. A., & Suarez, A. V. (2021). Pervasive and persistent effects of ant invasion and fragmentation on native ant assemblages. *Ecology, 102*(3). https://doi.org/10.1002/ecy.3257

Adenle, A. A., Stevens, C., & Bridgewater, P. (2015). Global conservation and management of biodiversity in developing countries: An opportunity for a new approach. *Environmental Science & Policy, 45*, 104–108. https://doi.org/10.1016/j.envsci.2014.10.002

Agamuthu, P., Mehran, S., Norkhairah, A., & Norkhairiyah, A. (2019). Marine debris: A review of impacts and global initiatives. *Waste Management & Research, 37*(10), 987–1002. https://doi.org/10.1177/0734242X19845041

Aguirre-Acosta, N., Kowaljow, E., & Aguilar, R. (2014). Reproductive performance of the invasive tree *Ligustrum lucidum* in a subtropical dry forest: Does habitat fragmentation boost or limit invasion? *Biological Invasions, 16*(7), 1397–1410. https://doi.org/10.1007/s10530-013-0577-x

Aieta, A., & Oliveira, K. (2009). Distribution, prevalence, and intensity of the swim bladder parasite *Anguillicola crassus* in New England and eastern Canada. *Diseases of Aquatic Organisms, 84*, 229–235. https://doi.org/10.3354/dao02049

Airoldi, L., Turon, X., Perkol-Finkel, S., & Rius, M. (2015). Corridors for aliens but not for natives: Effects of marine urban sprawl at a regional scale. *Diversity and Distributions, 21*(7), 755–768. https://doi.org/10.1111/ddi.12301

Aizen, M. A., Morales, C. L., Vázquez, D. P., Garibaldi, L. A., Sáez, A., & Harder, L. D. (2014). When mutualism goes bad: Density-dependent impacts of introduced bees on plant reproduction. *New Phytologist, 204*(2), 322–328. https://doi.org/10.1111/nph.12924

Aizen, M. A., Smith-Ramírez, C., Morales, C. L., Vieli, L., Sáez, A., Barahona-Segovia, R. M., Arbetman, M. P., Montalva, J., Garibaldi, L. A., Inouye, D. W., & Harder, L. D. (2019). Coordinated species importation policies are needed to reduce serious invasions globally: The case of alien bumblebees in South America. *Journal of Applied Ecology, 56*(1), 100–106. https://doi.org/10.1111/1365-2664.13121

Aketarawong, N., Isasawin, S., Sojikul, P., & Thanaphum, S. (2015). Gene flow and genetic structure of *Bactrocera carambolae* (Diptera, Tephritidae) among geographical differences and sister species, *B. dorsalis*, inferred from microsatellite DNA data. *ZooKeys, 540*, 239–272. https://doi.org/10.3897/zookeys.540.10058

Alahuhta, J., Erős, T., Kärnä, O. M., Soininen, J., JianJun, W., & Heino, J. (2019). Understanding environmental change through the lens of trait-based, functional, and phylogenetic biodiversity in freshwater ecosystems. *Environmental Reviews, 27*(2), Article 2. https://doi.org/10.1139/er-2018-0071

Al-Assaf, A., Tadros, M. J., Al-Shishany, S., Stewart, S., Majdalawi, M., Tabieh, M., & Othman, Y. A. (2020). Economic Assessment and Community Management of *Prosopis juliflora* Invasion in Sweimeh Village, Jordan. *Sustainability, 12*(20), Article 20. https://doi.org/10.3390/su12208327

Alba, C., Skálová, H., McGregor, K. F., D'Antonio, C., & Pyšek, P. (2015). Native and exotic plant species respond differently to wildfire and prescribed fire as revealed by meta-analysis. *Journal of Vegetation Science, 26*(1), 102–113. https://doi.org/10.1111/jvs.12212

Albuquerque, F. S., Macías-Rodríguez, M. Á., Búrquez, A., & Astudillo-Scalia, Y. (2019). Climate change and the potential expansion of buffelgrass (*Cenchrus ciliaris* L., Poaceae) in biotic communities of Southwest United States and northern Mexico. *Biological Invasions, 21*(11), 3335–3347. https://doi.org/10.1007/s10530-019-02050-5

Alexander, J. M., Kueffer, C., Daehler, C. C., Edwards, P. J., Pauchard, A., Seipel, T., & MIREN Consortium. (2011). Assembly of nonnative floras along elevational gradients explained by directional ecological filtering. *Proceedings of the National Academy of Sciences, 108*(2), 656–661. https://doi.org/10.1073/pnas.1013136108

Alexander, J. M., Lembrechts, J. J., Cavieres, L. A., Daehler, C., Haider, S., Kueffer, C., Liu, G., McDougall, K., Milbau, A., Pauchard, A., Rew, L. J., & Seipel, T. (2016). Plant invasions into mountains and alpine ecosystems: Current status and future challenges. *Alpine Botany, 126*(2), 89–103. https://doi.org/10.1007/s00035-016-0172-8

Ali, S. H., Giurco, D., Arndt, N., Nickless, E., Brown, G., Demetriades, A., Durrheim, R., Enriquez, M. A., Kinnaird, J., Littleboy, A., Meinert, L. D., Oberhänsli, R., Salem, J., Schodde, R., Schneider, G., Vidal, O., & Yakovleva, N. (2017). Mineral supply for sustainable development requires resource governance. *Nature, 543*(7645), Article 7645. https://doi.org/10.1038/nature21359

Aliani, S., & Molcard, A. (2003). Hitch-hiking on floating marine debris: Macrobenthic species in the Western Mediterranean Sea. *Hydrobiologia, 503*(1–3), 59–67. https://doi.org/10.1023/B:HYDR.0000008480.95045.26

Allen, J. M., Leininger, T. J., Hurd, J. D., Civco, D. L., Gelfand, A. E., & Silander, J. A. (2013). Socioeconomics drive woody invasive plant richness in New England, USA through forest fragmentation. *Landscape Ecology, 28*(9), 1671–1686. https://doi.org/10.1007/s10980-013-9916-7

Alofs, K. M., & Jackson, D. A. (2014). Meta-analysis suggests biotic resistance in freshwater environments is driven by consumption rather than competition. *Ecology, 95*(12), 3259–3270. https://doi.org/10.1890/14-0060.1

Aloo, P. A., Njiru, J., Balirwa, J. S., & Nyamweya, C. S. (2017). Impacts of Nile Perch, *Lates niloticus*, introduction on the ecology, economy and conservation of Lake Victoria, East Africa. *Lakes & Reservoirs: Science, Policy and Management for Sustainable Use, 22*(4), 320–333. https://doi.org/10.1111/lre.12192

Altamirano, A., Cely, J. P., Etter, A., Miranda, A., Fuentes-Ramirez, A., Acevedo, P., Salas, C., & Vargas, R. (2016). The invasive species *Ulex europaeus* (Fabaceae) shows high dynamism in a fragmented landscape of south-central Chile. *Environmental Monitoring and Assessment, 188*(8), 495. https://doi.org/10.1007/s10661-016-5498-6

Amanor, K. S. (1991). Managing the Fallow: Weeding Technology and Environmental Knowledge in the Krobo District of Ghana. *Agriculture and Human Values, 8*(1–2), 5–13. https://doi.org/10.1007/BF01579651

An, S. Q., Gu, B. H., Zhou, C. F., Wang, Z. S., Deng, Z. F., Zhi, Y. B., Li, H. L., Chen, L., Yu, D. H., & Liu, Y. H. (2007). *Spartina* invasion in China: Implications for invasive species management and future research. *Weed Research, 47*(3), 183–191. https://doi.org/10.1111/j.1365-3180.2007.00559.x

Anderson, L. G., Rocliffe, S., Haddaway, N. R., & Dunn, A. M. (2015). The Role of Tourism and Recreation in the Spread of Non-Native Species: A Systematic Review and Meta-Analysis. *PLoS ONE, 10*(10), e0140833. https://doi.org/10.1371/journal.pone.0140833

Anderson, P. K., Cunningham, A. A., Patel, N. G., Morales, F. J., Epstein, P. R., & Daszak, P. (2004). Emerging infectious diseases of plants: Pathogen pollution, climate change and agrotechnology drivers. *Trends in Ecology & Evolution, 19*(10), 535–544. https://doi.org/10.1016/j.tree.2004.07.021

Andraca-Gómez, G., Ordano, M., Boege, K., Domínguez, C. A., Piñero, D., Pérez-Ishiwara, Pérez-Camacho, J., Cañizares, M., & Fornoni, J. (2015). A potential invasion route of *Cactoblastis cactorum* within the Caribbean region matches historical hurricane trajectories. *Biological Invasions, 17*(5), 1397–1406. https://doi.org/10.1007/s10530-014-0802-2

Andrews, J. D. (1984). Epizootiology of diseases of oysters (*Crassostrea virginica*), and parasites of associated organisms in eastern North America. *Helgoländer Meeresuntersuchungen, 37*(149–166). https://doi.org/10.1007/BF01989300

Andriantsoa, R., Jones, J. P. G., Achimescu, V., Randrianarison, H., Raselimanana, M., Andriatsitohaina, M., Rasamy, J., & Lyko, F. (2020). Perceived socio-economic impacts of the marbled crayfish invasion in Madagascar. *PLOS ONE, 15*(4), e0231773. https://doi.org/10.1371/journal.pone.0231773

Angelo, C. L., & Daehler, C. C. (2013). Upward expansion of fire-adapted grasses along a warming tropical elevation gradient. *Ecography, 36*(5), 551–559. https://doi.org/10.1111/j.1600-0587.2012.07754.x

Anufriieva, E. V., & Shadrin, N. V. (2018). Extreme hydrological events destabilize aquatic ecosystems and open doors for alien species. *Quaternary International, 475*, 11–15. https://doi.org/10.1016/j.quaint.2017.12.006

Arbetman, M. P., Meeus, I., Morales, C. L., Aizen, M. A., & Smagghe, G. (2013). Alien parasite hitchhikes to Patagonia on invasive bumblebee. *Biological Invasions, 15*(3), 489–494. https://doi.org/10.1007/s10530-012-0311-0

Arcella, T. E., Perry, W. L., Lodge, D. M., & Feder, J. L. (2014). The role of hybridization in a species invasion and extirpation of resident fauna: Hybrid vigor and breakdown in the rusty crayfish, Orconectes rusticus. *Journal of Crustacean Biology, 34*(2), 157–164. https://doi.org/10.1163/1937240X-00002204

Arechavala-Lopez, P., Toledo-Guedes, K., Izquierdo-Gomez, D., Šegvić-Bubić, T., & Sanchez-Jerez, P. (2018). Implications of Sea Bream and Sea Bass Escapes for Sustainable Aquaculture Management: A Review of Interactions, Risks and Consequences. *Reviews in Fisheries Science & Aquaculture, 26*(2), 214–234. https://doi.org/10.1080/23308249.2017.1384789

Arechavala-Lopez, P., Uglem, I., Izquierdo-Gomez, D., Fernandez-Jover, D., & Sanchez-Jerez, P. (2017). Rapid dispersion of escaped meagre *(Argyrosomus regius)* from a coastal Mediterranean fish farm. *Aquaculture Research, 48*(4), 1502–1512. https://doi.org/10.1111/are.12986

Arias, P., Bellouin, N., Coppola, E., Jones, R., Krinner, G., Marotzke, J., Naik, V., Palmer, M., Plattner, G.-K., & Rogelj, J. (2021). *Working Group 1: The Physical Science Basis; Technical Summary.* Sixth Assessment Report of the Intergovernmental Panel on Climate Change. https://www.ipcc.ch/report/ar6/wg1/chapter/technical-summary/

Armitage, J., Edwards, D., Konyves, K., Lancaster, N., Marshall, R., Cubey, J., & Merrick, J. (2016). *RHS plant finder*. Wisley: Royal Horticultural Society.

Ashton, G. V., Willis, K. J., Burrows, M. T., & Cook, E. J. (2007). Environmental tolerance of *Caprella mutica*: Implications for its distribution as a marine non-native species. *Marine Environmental Research, 64*(3), 305–312. https://doi.org/10.1016/j.marenvres.2007.02.003

Aslan, C. E., & Dickson, B. G. (2020). Non-native plants exert strong but under-studied influence on fire dynamics. *NeoBiota, 61*, 47–64. https://doi.org/10.3897/neobiota.61.51141

Asth, M. da S., Rodrigues, R. G., & Zenni, R. D. (2021). Canals as invasion pathways in tropical dry forest and the need for monitoring and management. *Journal of Applied Ecology, 58*(9), 2004–2014. https://doi.org/10.1111/1365-2664.13950

Atalah, J., Hopkins, G. A., & Forrest, B. M. (2013). Augmentative Biocontrol in Natural Marine Habitats: Persistence, Spread and Non-Target Effects of the Sea Urchin *Evechinus chloroticus. PLoS ONE, 8*(11), e80365. https://doi.org/10.1371/journal.pone.0080365

Australian Government. (2021). *Myrtle rust* (Austropuccinia psidii). Department of Climate Change, Energy, the Environment and Water. https://www.dcceew.gov.au/environment/invasive-species/diseases-fungi-and-parasites/myrtle-rust

Avio, C. G., Gorbi, S., & Regoli, F. (2017). Plastics and microplastics in the oceans: From emerging pollutants to emerged threat. *Marine Environmental Research, 128*, 2–11. https://doi.org/10.1016/j.marenvres.2016.05.012

Aylor, D. E. (2003). Spread of plant disease on a continental scale: Role of aerial dispersal of pathogens. *Ecology, 84*(8), 1989–1997. https://www.jstor.org/stable/3450023?seq=1&cid=pdf-reference#references_tab_contents

Azevedo-Santos, V. M., Vitule, J. R. S., Pelicice, F. M., García-Berthou, E., & Simberloff, D. (2017). Nonnative Fish to Control *Aedes* Mosquitoes: A Controversial, Harmful Tool. *BioScience, 67*(1), 84–90. https://doi.org/10.1093/biosci/biw156

Bacon, S. J., Bacher, S., & Aebi, A. (2012). Gaps in Border Controls Are Related to Quarantine Alien Insect Invasions in Europe. *PLoS ONE, 7*(10), e47689. https://doi.org/10.1371/journal.pone.0047689

Bailey, S. A., Brown, L., Campbell, M. L., Canning-Clode, J., Carlton, J. T., Castro, N., Chainho, P., Chan, F. T., Creed, J. C., Curd, A., Darling, J., Fofonoff, P., Galil, B. S., Hewitt, C. L., Inglis, G. J., Keith, I., Mandrak, N. E., Marchini, A., McKenzie, C. H., … Zhan, A. (2020). Trends in the detection of aquatic non-indigenous species across global marine, estuarine and freshwater ecosystems: A 50-year perspective. *Diversity and Distributions, 26*(12), 1780–1797. https://doi.org/10.1111/ddi.13167

Baker, R. H. A., Cannon, R. J. C., & Walters, K. F. A. (1996). An assessment of the risks posed by selected non-indigenous pests to UK crops under climate change. *Aspects of Applied Biology,* 323–330. https://www.cabdirect.org/cabdirect/abstract/19961105741

Balch, J. K., Bradley, B. A., D'Antonio, C. M., & Gómez-Dans, J. (2013). Introduced annual grass increases regional fire activity across the arid western USA (1980-2009). *Global Change Biology, 19*(1), 173–183. https://doi.org/10.1111/gcb.12046

Baldacchino, F., Caputo, B., Chandre, F., Drago, A., della Torre, A., Montarsi, F., & Rizzoli, A. (2015). Control methods against invasive *Aedes* mosquitoes in Europe: A review. *Pest Management Science, 71*(11), 1471–1485. https://doi.org/10.1002/ps.4044

Bale, J. S., Masters, G. J., Hodkinson, I. D., Awmack, C., Bezemer, T. M., Brown, V. K., Butterfield, J., Buse, A., Coulson, J. C., Farrar, J., Good, J. E. G., Harrington, R., Hartley, S., Jones, T. H., Lindroth, R. L., Press, M. C., Symrnioudis, I., Watt, A. D., & Whittaker, J. B. (2002). Herbivory in global climate change research: Direct effects of rising temperature on insect herbivores. *Global Change Biology, 8*(1), 1–16. https://doi.org/10.1046/j.1365-2486.2002.00451.x

Banks, N. C., Paini, D. R., Bayliss, K. L., & Hodda, M. (2015). The role of global trade and transport network topology in the human-mediated dispersal of alien species. *Ecology Letters, 18*(2), 188–199. https://doi.org/10.1111/ele.12397

Barbosa, N. P. U., Wilson Fernandes, G., Carneiro, M. A. A., & Júnior, L. A. C. (2010). Distribution of non-native invasive species and soil properties in proximity to paved roads and unpaved roads in a quartzitic mountainous grassland of southeastern Brazil (rupestrian fields). *Biological Invasions, 12*(11), 3745–3755. https://doi.org/10.1007/s10530-010-9767-y

Barnes, T. G., DeMaso, S. J., & Bahm, M. A. (2013). The Impact of 3 exotic, invasive grasses in the Southeastern United States on wildlife: Exotic Grass Management in the Southeast. *Wildlife Society Bulletin*, 497–502. https://doi.org/10.1002/wsb.316

Barney, J. N., & DiTomaso, J. M. (2008). Nonnative Species and Bioenergy: Are We Cultivating the Next Invader? *BioScience, 58*(1), Article 1. https://doi.org/10.1641/B580111

Barrat, J., Richomme, C., & Moinet, M. (2010). The accidental release of exotic species from breeding colonies and zoological collections: -EN- -FR- Lâcher accidentel d'espèces exotiques à partir de colonies d'élevage et de collections zoologiques -ES- Liberación accidental de especies exóticas de colonias reproductoras y colecciones zoológicas. *Revue Scientifique et Technique de l'OIE, 29*(1), 113–122. https://doi.org/10.20506/rst.29.1.1968

Barrios, J., Bolaños, J., & López, R. (2007). Blanqueamiento de arrecifes coralinos por la invasión de *Kappaphycus alvarezii* (rhodophyta) en isla cubagua, estado nueva esparta, Venezuela. *Boletín Del Instituto Oceanográfico de Venezuela, 46*(2), 147–152. https://repositorioslatinoamericanos.uchile.cl/handle/2250/227223

Barrios-Garcia, M. N., & Ballari, S. A. (2012). Impact of wild boar *(Sus scrofa)* in its introduced and native range: A review. *Biological Invasions, 14*(11), Article 11. https://doi.org/10.1007/s10530-012-0229-6

Barrios-O'Neill, D., Kelly, R., Dick, J. T. A., Ricciardi, A., MacIsaac, H. J., & Emmerson, M. C. (2016). On the context-dependent scaling of consumer feeding rates. *Ecology Letters, 19*(6), 668–678. https://doi.org/10.1111/ele.12605

Barse, A. M., McGuire, S. A., Vinores, M. A., Eierman, L. E., & Weeder, J. A. (2001). The swimbladder nematode *Anguillicola crassus* in american eels *(Anguilla rostrata)* from middle and upper regions of Chesapeake Bay. *Journal of Parasitology, 87*(6), 1366–1370. https://doi.org/10.1645/0022-3395(2001)087[1366:TSNACI]2.0.CO;2

Bartley, D. M., & Bell, J. D. (2008). Restocking, Stock Enhancement, and Sea Ranching: Arenas of Progress. *Reviews in Fisheries Science, 16*(1–3), 357–365. https://doi.org/10.1080/10641260701678058

Bartomeus, I., Ascher, J. S., Gibbs, J., Danforth, B. N., Wagner, D. L., Hedtke, S. M., & Winfree, R. (2013). Historical changes in northeastern US bee pollinators related to shared ecological traits. *Proceedings of the National Academy of Sciences, 110*(12), 4656–4660. https://doi.org/10.1073/pnas.1218503110

Batanouny, K. H., & El-Fiky, A. M. (1984). Water hyacinth in Egypt: Distribution and problem magnitude. *Proceedings of the International Conference on Water Hyacinth : Hyderabad, India, February 7-11, 1983 / Editor: G. Thyagarajan*.http://agris.fao.org/agris-search/search.do?recordID=US201301417304

Bažok, R., Lemić, D., Chiarini, F., & Furlan, L. (2021). Western Corn Rootworm (*Diabrotica virgifera virgifera* LeConte) in Europe: Current Status and Sustainable Pest Management. *Insects, 12*(3), 195. https://doi.org/10.3390/insects12030195

Becker, M., Alvarez, M., Heller, G., Leparmarai, P., Maina, D., Malombe, I., Bollig, M., & Vehrs, H. (2016). Land-use changes and the invasion dynamics of shrubs in Baringo. *Journal of Eastern African Studies, 10*(1), 111–129. https://doi.org/10.1080/17531055.2016.1138664

Becker, T., Dietz, H., Billeter, R., Buschmann, H., & Edwards, P. J. (2005). Altitudinal distribution of alien plant species in the Swiss Alps. *Perspectives in Plant Ecology, Evolution and Systematics, 7*(3), Article 3. https://doi.org/10.1016/j.ppees.2005.09.006

Bekele, K., Haji, J., Legesse, B., Shiferaw, H., & Schaffner, U. (2018). Impacts of woody invasive alien plant species on rural livelihood: Generalized propensity score evidence from *Prosopis* spp. invasion in Afar Region in Ethiopia. *Pastoralism: Research, Policy and Practice, 8*(1), 28. https://doi.org/10.1186/s13570-018-0124-6

Bell, D. J., & Webb, N. J. (1991). Effects of climate on reproduction in the European wild rabbit *(Oryctolugus cuniculus)*. *Journal of Zoology, 224*(4), 639–648. https://doi.org/10.1111/j.1469-7998.1991.tb03792.x

Bell, F., & Donnelly, L. (2006). *Mining and its Impact on the Environment.* CRC Press, Boca Raton, FL. https://www.crcpress.com/Mining-and-its-Impact-on-the-Environment/Bell-Donnelly/p/book/9780367390792

Bellard, C., Leroy, B., Thuiller, W., Rysman, J.-F., & Courchamp, F. (2016). Major drivers of invasion risks throughout the world. *Ecosphere, 7*(3), e01241. https://doi.org/10.1002/ecs2.1241

Bellard, C., Thuiller, W., Leroy, B., Genovesi, P., Bakkenes, M., & Courchamp, F. (2013). Will climate change promote future invasions? *Global Change Biology, 19*(12), 3740–3748. https://doi.org/10.1111/gcb.12344

Bellingham, P. J., Tanner, E. V. J., & Healey, J. R. (2005). Hurricane disturbance accelerates invasion by the alien tree Pittosporum undulatum in Jamaican montane rain forests. *Journal of Vegetation Science, 16*(6), 675–684.

Beltrán-Beck, B., García, F. J., & Gortázar, C. (2012). Raccoons in Europe: Disease hazards due to the establishment of an invasive species. *European Journal of Wildlife Research, 58*(1), 5–15. https://doi.org/10.1007/s10344-011-0600-4

Benedict, M. Q., Levine, R. S., Hawley, W. A., & Lounibos, L. P. (2007). Spread of the tiger: Global risk of invasion by the mosquito *Aedes albopictus. Vector-Borne and Zoonotic Diseases, 7*(1), 76–85. https://doi.org/10.1089/vbz.2006.0562

Bennett, B. M. (2010). The El Dorado of Forestry: The Eucalyptus in India, South Africa, and Thailand, 1850–2000. *International Review of Social History, 55*(S18), 27–50. https://doi.org/10.1017/S0020859010000489

Bennett, B. M. (2011). Naturalising Australian Trees in South Africa: Climate, Exotics and Experimentation. *Journal of Southern African Studies, 37*(2), Article 2. https://doi.org/10.1080/03057070.2011.579434

Bergquist, E., Evangelista, P., Stohlgren, T. J., & Alley, N. (2007). Invasive Species and Coal Bed Methane Development in the Powder River Basin, Wyoming. *Environmental Monitoring and Assessment, 128*(1–3), 381–394. https://doi.org/10.1007/s10661-006-9321-7

Berntsen, H. H., Sandlund, O. T., & Thorstad, E. B. (2022). Pukkellaks i Norge 2021. In *34*. Norsk institutt for naturforskning (NINA). https://brage.nina.no/nina-xmlui/handle/11250/3018858

Bertelsmeier, C., Ollier, S., Liebhold, A., & Keller, L. (2017a). Recent human history governs global ant invasion dynamics. *Nature Ecology & Evolution, 1*(7), 1–8. https://doi.org/10.1038/s41559-017-0184

Bhattarai, G. P., & Cronin, J. T. (2014). Hurricane Activity and the Large-Scale Pattern of Spread of an Invasive Plant Species. *PLoS ONE, 9*(5), e98478. https://doi.org/10.1371/journal.pone.0098478

Bij de Vaate, A., Jazdzewski, K., Ketelaars, H. A. M., Gollasch, S., & Van der Velde, G. (2002). Geographical patterns in range extension of Ponto-Caspian macroinvertebrate species in Europe. *Canadian Journal of Fisheries and Aquatic Sciences, 59*(7), 1159–1174. https://doi.org/10.1139/f02-098

Biton, E. (2020). Possible implications of sea level changes for species migration through the Suez Canal. *Scientific Reports, 10*(1), 21195. https://doi.org/10.1038/s41598-020-78313-2

Blackburn, T. M., Pyšek, P., Bacher, S., Carlton, J. T., Duncan, R. P., Jarošík, V., Wilson, J. R. U., & Richardson, D. M. (2011). A proposed unified framework for biological invasions. *Trends in Ecology & Evolution, 26*(7), 333–339. https://doi.org/10.1016/j.tree.2011.03.023

Blanco-Aguiar, J. A., González-Jara, P., Ferrero, M. E., Sánchez-Barbudo, I., Virgós, E., Villafuerte, R., & Dávila, J. A. (2008). Assessment of game restocking contributions to anthropogenic hybridization: The case of the Iberian red-legged partridge. *Animal Conservation, 11*(6), 535–545. https://doi.org/10.1111/j.1469-1795.2008.00212.x

Boer, M. M., Dios, V. R. de, & Bradstock, R. A. (2020). Unprecedented burn area of Australian mega forest fires. *Nature Climate Change, 10*(3), Article 3. https://doi.org/10.1038/s41558-020-0716-1

Bolstad, G. H., Hindar, K., Robertsen, G., Jonsson, B., Sægrov, H., Diserud, O. H., Fiske, P., Jensen, A. J., Urdal, K., Næsje, T. F., Barlaup, B. T., Florø-Larsen, B., Lo, H., Niemelä, E., & Karlsson, S. (2017). Gene flow from domesticated escapes alters the life history of wild Atlantic salmon. *Nature Ecology & Evolution, 1*(5), 124. https://doi.org/10.1038/s41559-017-0124

Bond, W. J., & Midgley, G. F. (2000). A proposed CO_2-controlled mechanism of woody plant invasion in grasslands and savannas. *Global Change Biology, 6*(8), 865–869. https://doi.org/10.1046/j.1365-2486.2000.00365.x

Bonebrake, T. C., Guo, F., Dingle, C., Baker, D. M., Kitching, R. L., & Ashton, L. A. (2019). Integrating Proximal and Horizon Threats to Biodiversity for Conservation. *Trends in Ecology & Evolution, 34*(9), 781–788. https://doi.org/10.1016/j.tree.2019.04.001

Bonesi, L., & Palazon, S. (2007). The American mink in Europe: Status, impacts, and control. *Biological Conservation, 134*(4), 470–483. https://doi.org/10.1016/j.biocon.2006.09.006

Borer, E. T., Grace, J. B., Harpole, W. S., MacDougall, A. S., & Seabloom, E. W. (2017). A decade of insights into grassland ecosystem responses to global environmental change. *Nature Ecology & Evolution, 1*(5), Article 5. https://doi.org/10.1038/s41559-017-0118

Boudouresque, C.-F., Blanfuné, A., Pergent, G., & Thibaut, T. (2021). Restoration of Seagrass Meadows in the Mediterranean Sea: A Critical Review of Effectiveness and Ethical Issues. *Water, 13*(8), 1034. https://doi.org/10.3390/w13081034

Boudouresque, C.-F., & Verlaque, M. (2012). An overview of species introduction and invasion processes in marine and coastal lagoon habitats. *Cahiers de Biologie Marine, 53*, 309–317.

Bowler, D. E., Bjorkman, A. D., Dornelas, M., Myers-Smith, I. H., Navarro, L. M., Niamir, A., Supp, S. R., Waldock, C., Winter, M., Vellend, M., Blowes, S. A., Böhning-Gaese, K., Bruelheide, H., Elahi, R., Antão, L. H., Hines, J., Isbell, F., Jones, H. P., Magurran, A. E., … Bates, A. E. (2020). Mapping human pressures on biodiversity across the planet uncovers anthropogenic threat complexes. *People and Nature, 2*(2), 380–394. https://doi.org/10.1002/pan3.10071

Bradley, B. A., Blumenthal, D. M., Early, R., Grosholz, E. D., Lawler, J. J., Miller, L. P., Sorte, C. J. B., D'Antonio, C. M., Diez, J. M., Dukes, J. S., Ibanez, I., & Olden, J. D. (2012). Global change, global trade, and the next wave of plant invasions. *Frontiers in Ecology and the Environment, 10*(1), 20–28. https://doi.org/10.1890/110145

Bradley, B. A., Blumenthal, D. M., Wilcove, D. S., & Ziska, L. H. (2010). Predicting plant invasions in an era of global change. *Trends in Ecology & Evolution, 25*(5), 310–318. https://doi.org/10.1016/j.tree.2009.12.003

Braga, R. R., Gómez Aparicio, L., Heger, T., Vitule, J. R. S., Jeschke, J. M., Jeschke, J. M., & Heger, T. (2018). Invasional meltdown hypothesis. *Invasion Biology: Hypotheses and Evidence,* 79–91.

Braga, R. R., Gómez-Aparicio, L., Heger, T., Vitule, J. R. S., & Jeschke, J. M. (2018). Structuring evidence for invasional meltdown: Broad support but with biases and gaps. *Biological Invasions, 20*(4), 923–936. https://doi.org/10.1007/s10530-017-1582-2

Braimah, H., & Timbilla, J. A. (2002). A decade of successful biological control of Siam weed, Chromolaena odorata in Ghana: Lessons and future plans. *Proceedings of the Fifth International Workshop on the Biological Control and Management of Chromolaena Odorata*, 58–65.

Brasier, C. M., & Buck, K. W. (2001). Rapid Evolutionary Changes in a Globally Invading Fungal Pathogen (Dutch Elm Disease).

Biological Invasions, 3(3), 223–233. https://doi.org/10.1023/A:1015248819864

Bridgman, L. J., Benitez, V. V., Graña Grilli, M., Mufato, N., Acosta, D., & Guichón, M. L. (2012). Short perceptual range and yet successful invasion of a fragmented landscape: The case of the red-bellied tree squirrel *(Callosciurus erythraeus)* in Argentina. *Landscape Ecology, 27*(5), 633–640. https://doi.org/10.1007/s10980-012-9727-2

Britton, J. R., & Orsi, M. L. (2012). Non-native fish in aquaculture and sport fishing in Brazil: Economic benefits *versus* risks to fish diversity in the upper River Paraná Basin. *Reviews in Fish Biology and Fisheries, 22*(3), 555–565. https://doi.org/10.1007/s11160-012-9254-x

Brodie, J., Williamson, C. J., Smale, D. A., Kamenos, N. A., Mieszkowska, N., Santos, R., Cunliffe, M., Steinke, M., Yesson, C., Anderson, K. M., Asnaghi, V., Brownlee, C., Burdett, H. L., Burrows, M. T., Collins, S., Donohue, P. J. C., Harvey, B., Foggo, A., Noisette, F., … Hall-Spencer, J. M. (2014). The future of the northeast Atlantic benthic flora in a high CO_2 world. *Ecology and Evolution, 4*(13), 2787–2798. https://doi.org/10.1002/ece3.1105

Brondizio, E., Diaz, S., Settele, J., Ngo, H. T., Gueze, M., Aumeeruddy-Thomas, Y., Bai, X., Geschke, A., Molnár, Z., Niamir, A., Pascual, U., Simcock, A., & Jaureguiberry, P. (2019). IPBES Global Assessment Report on Biodiversity and Ecosystem Services Chapter 1 Assessing a planet in transformation: Rationale and approach. In *IPBES Global Assessment on Biodiversity and Ecosystem Services.* Zenodo. https://doi.org/10.5281/ZENODO.3831852

Brooks, M. L. (2003). Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. *Journal of Applied Ecology, 40*(2), 344–353. https://doi.org/10.1046/j.1365-2664.2003.00789.x

Brooks, M. L., D'Antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., Hobbs, R. J., Pellant, M., & Pyke, D. (2004). Effects of Invasive Alien Plants on Fire Regimes. *BioScience, 54*(7), Article 7. https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2

Brown, C. M. (2008). Reaping the Whirlwind? Human Disease from Exotic Pets. *BioScience, 58*(1), 6–7. https://doi.org/10.1641/B580102

Brown, E. B. E., Adkin, A., Fooks, A. R., Stephenson, B., Medlock, J. M., & Snary, E. L. (2012). Assessing the Risks of West Nile Virus-Infected Mosquitoes from Transatlantic Aircraft: Implications for Disease Emergence in the United Kingdom. Vector-Borne and Zoonotic Diseases, 12(4), 310–320. https://doi.org/10.1089/vbz.2010.0176

Brown, J. K. M., & Hovmøller, M. S. (2002). Aerial dispersal of pathogens on the global and continental scales and its impact on plant disease. *Science, 297*(5581), 537–541. https://doi.org/10.1126/science.1072678

Brown, M. E., & Walls, S. C. (2013). Variation in Salinity Tolerance among Larval Anurans: Implications for Community Composition and the Spread of an Invasive, Non-native Species. *Copeia, 2013*(3), 543–551. https://doi.org/10.1643/CH-12-159

Brummer, T. J., Byrom, A. E., Sullivan, J. J., & Hulme, P. E. (2016). Alien and native plant richness and abundance respond to different environmental drivers across multiple gravel floodplain ecosystems. *Diversity and Distributions, 22*(7), 823–835. https://doi.org/10.1111/ddi.12448

Brundu, G. (2015). Plant invaders in European and Mediterranean inland waters: Profiles, distribution, and threats. *Hydrobiologia, 746*(1), 61–79. https://doi.org/10.1007/s10750-014-1910-9

Brundu, G., Pauchard, A., Pyšek, P., Pergl, J., Bindewald, A., Brunori, A., Canavan, S., Campagnaro, T., Celesti-Grapow, L., de Sá Dechoum, M., Dufour-Dror, J.-M., Essl, F., Flory, S. L., Genovesi, P., Guarino, F., Guangzhe, L., Hulme, P. E., Jäger, H., Kettle, C. J., … Richardson, D. M. (2020). Global guidelines for the sustainable use of non-native trees to prevent tree invasions and mitigate their negative impacts. *NeoBiota, 61*, 65–116. https://doi.org/10.3897/neobiota.61.58380

Brzuszek, R. F., & Harkess, R. L. (2009). Green Industry Survey of Native Plant Marketing in the Southeastern United States. *HortTechnology, 19*(1), Article 1. https://doi.org/10.21273/HORTSCI.19.1.168

Bulleri, F., & Airoldi, L. (2005). Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *Tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology, 42*(6), 1063–1072. https://doi.org/10.1111/j.1365-2664.2005.01096.x

Bulleri, F., & Chapman, M. G. (2010). The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology, 47*, 26–35. https://doi.org/10.1111/j.1365-2664.2009.01751.x

Burgess, T. I., Crous, C. J., Slippers, B., Hantula, J., & Wingfield, M. J. (2016). Tree invasions and biosecurity: Eco-evolutionary dynamics of hitchhiking fungi. *AoB PLANTS, 8*, plw076. https://doi.org/10.1093/aobpla/plw076

Burreson, E. M., Stokes, N. A., & Friedman, C. S. (2000). Increased Virulence in an Introduced Pathogen: *Haplosporidium nelsoni* (MSX) in the Eastern Oyster *Crassostrea virginica*. *Journal of Aquatic Animal Health, 12*(1), 1–8. https://doi.org/10.1577/1548-8667(2000)012<0001:IVIAIP>2.0.CO;2

Burrows, N. D. (2018). Feral Animals in the Semi-arid and Arid Regions of Australia: Origins, Impacts and Control. In H. Lambers (Ed.), *On the Ecology of Australia's Arid Zone* (pp. 331–373). Springer International Publishing. https://doi.org/10.1007/978-3-319-93943-8_13

Butler, D. W., & Fairfax, R. J. (2003). Buffel Grass and fire in a Gidgee and Brigalow woodland: A case study from central Queensland. *Ecological Management & Restoration, 4*(2), 120–125. https://doi.org/10.1046/j.1442-8903.2003.00146.x

Butler, S., McAlpine, C., Fensham, R., & House, A. (2014). Climate and exotic pasture area in landscape determines invasion of forest fragments by two invasive grasses. *Journal of Applied Ecology, 51*(1), 114–123. https://doi.org/10.1111/1365-2664.12160

Butt, N., Beyer, H. L., Bennett, J. R., Biggs, D., Maggini, R., Mills, M., Renwick, A. R., Seabrook, L. M., & Possingham, H. P. (2013). Biodiversity Risks from Fossil Fuel Extraction. *Science, 342*(6157), 425–426. https://doi.org/10.1126/science.1237261

Bwanika, G. N., Chapman, L. J., Kizito, Y., & Balirwa, J. (2006). Cascading effects of introduced Nile perch *(Lates niloticus)* on the foraging ecology of Nile tilapia *(Oreochromis niloticus)*. *Ecology of Freshwater Fish, 15*(4), 470–481. https://doi.org/10.1111/j.1600-0633.2006.00185.x

Byun, C., de Blois, S., & Brisson, J. (2015). Interactions between abiotic constraint, propagule pressure, and biotic resistance regulate plant invasion. *Oecologia, 178*(1), 285–296. https://doi.org/10.1007/s00442-014-3188-z

Caley, P., Ingram, R., & De Barro, P. (2015). Entry of exotic insects into Australia: Does border interception count match incursion risk? *Biological Invasions, 17*(4), 1087–1094. https://doi.org/10.1007/s10530-014-0777-z

Callaham, M. A., & Blair, J. M. (1999). Influence of differing land management on the invasion of North American tallgrass prairie soils by European earthworms. *Pedobiologia, 43*, 507–512. http://lter.konza.ksu.edu/content/influence-differing-land-management-invasion-north-american-tallgrass-prairie-soils-european

Callaham, M. A., Blair, J. M., Todd, T. C., Kitchen, D. J., & Whiles, M. R. (2003). Macroinvertebrates in North American tallgrass prairie soils: Effects of fire, mowing, and fertilization on density and biomass. *Soil Biology and Biochemistry, 35*(8), 1079–1093. https://doi.org/10.1016/S0038-0717(03)00153-6

Callaham, M. A., Stanturf, J. A., Hammond, W. J., Rockwood, D. L., Wenk, E. S., & O'Brien, J. J. (2013). Survey to Evaluate Escape of *Eucalyptus* spp. Seedlings from Plantations in Southeastern USA. *International Journal of Forestry Research, 2013*, e946374. https://doi.org/10.1155/2013/946374

Calle, L., & Gawlik, D. E. (2011). Anthropogenic food in the diet of the Sacred Ibis *(Threskiornis aethiopicus)*, a non-native wading bird in southeastern Florida, USA. *Florida Field Naturalist, 39*(1), 1–34. https://www.researchgate.net/profile/Leonardo_Calle/publication/260263587_Anthropogenic_food_in_the_diet_of_the_Sacred_Ibis_Threskiornis_aethiopicus_a_non-native_wading_bird_in_southeastern_FL_USA/links/00b7d530612673274a000000/Anthropogenic-food-in-the-diet-of-the-Sacred-Ibis-Threskiornis-aethiopicus-a-non-native-wading-bird-in-southeastern-FL-USA.pdf

Camarero, P. (2019). Exotic vine invasions following cyclone disturbance in Australian Wet Tropics rainforests: A review. *Austral Ecology, 44*(8), 1359–1372. https://doi.org/10.1111/aec.12810

Cambray, J. A. (2003). Impact on indigenous species biodiversity caused by the globalisation of alien recreational freshwater fisheries. *Hydrobiologia, 500*(1), 217–230. https://doi.org/10.1023/A:1024648719995

Cameron, E. K., & Bayne, E. M. (2009). Road age and its importance in earthworm invasion of northern boreal forests. *Journal of Applied Ecology, 46*(1), 28–36. https://doi.org/10.1111/j.1365-2664.2008.01535.x

Campbell, M. L., King, S., Heppenstall, L. D., van Gool, E., Martin, R., & Hewitt, C. L. (2017). Aquaculture and urban marine structures facilitate native and non-indigenous species transfer through generation and accumulation of marine debris. *Marine Pollution Bulletin, 123*(1–2), 304–312. https://doi.org/10.1016/j.marpolbul.2017.08.040

Canonico, G. C., Arthington, A., McCrary, J. K., & Thieme, M. L. (2005). The effects of introduced tilapias on native biodiversity. *Aquatic Conservation: Marine and Freshwater Ecosystems, 15*(5), 463–483. https://doi.org/10.1002/aqc.699

Canu, A., Apollonio, M., & Scandura, M. (2018). Unmasking the invader: Genetic identity of invasive wild boar from three minor islands off Sardinia (Italy). *Mammalian Biology, 93*, 29–37. https://doi.org/10.1016/j.mambio.2018.07.008

Capel, K. C. C., Creed, J., Kitahara, M. V., Chen, C. A., & Zilberberg, C. (2019). Multiple introductions and secondary dispersion of *Tubastraea* spp. In the Southwestern Atlantic. *Scientific Reports, 9*(1), 13978. https://doi.org/10.1038/s41598-019-50442-3

Carlile, W. R. (1988). F*ungicides. Control of crop diseases.*

Carlton, J. T., Chapman, J. W., Geller, J. B., Miller, J. A., Carlton, D. A., McCuller, M. I., Treneman, N. C., Steves, B. P., & Ruiz, G. M. (2017). Tsunami-driven rafting: Transoceanic species dispersal and implications for marine biogeography. *Science, 357*(6358), 1402–1406. https://doi.org/10.1126/science.aao1498

Carpio, A. J., Álvarez, Y., Oteros, J., León, F., & Tortosa, F. S. (2020). Intentional introduction pathways of alien birds and mammals in Latin America. *Global Ecology and Conservation, 22*, e00949. https://doi.org/10.1016/j.gecco.2020.e00949

Carpio, A. J., De Miguel, R. J., Oteros, J., Hillström, L., & Tortosa, F. S. (2019). Angling as a source of non-native freshwater fish: A European review. *Biological Invasions, 21*(11), 3233–3248. https://doi.org/10.1007/s10530-019-02042-5

Carpio, A. J., Guerrero-Casado, J., Barasona, J. A., Tortosa, F. S., Vicente, J., Hillström, L., & Delibes-Mateos, M. (2017). Hunting as a source of alien species: A European review. *Biological Invasions, 19*(4), 1197–1211. https://doi.org/10.1007/s10530-016-1313-0

Casoli, E., Mancini, G., Ventura, D., Belluscio, A., & Ardizzone, G. (2021). Double Trouble: Synergy between Habitat Loss and the Spread of the Alien Species *Caulerpa cylindracea* (Sonder) in Three Mediterranean Habitats. *Water, 13*(10), 1342.

Catford, J. A., Daehler, C. C., Murphy, H. T., Sheppard, A. W., Hardesty, B. D., Westcott, D. A., Rejmánek, M., Bellingham, P. J., Pergl, J., Horvitz, C. C., & Hulme, P. E. (2012). The intermediate disturbance hypothesis and plant invasions: Implications for species richness and management. *Perspectives in Plant Ecology, Evolution and Systematics, 14*(3), 231–241. https://doi.org/10.1016/j.ppees.2011.12.002

Catford, J. A., Downes, B. J., Gippel, C. J., & Vesk, P. A. (2011). Flow regulation reduces native plant cover and facilitates exotic invasion in riparian wetlands. *Journal of Applied Ecology, 48*(2), 432–442. https://doi.org/10.1111/j.1365-2664.2010.01945.x

Catford, J. A., Morris, W. K., Vesk, P. A., Gippel, C. J., & Downes, B. J. (2014). Species and environmental characteristics point to flow regulation and drought as drivers of riparian plant invasion. *Diversity and Distributions, 20*(9), 1084–1096. https://doi.org/10.1111/ddi.12225

Caudell, J. N., Dowell, E., & Welch, K. (2016). Economic utility for the anthropogenic spread of wild hogs. *Human–Wildlife Interactions, 10*(2), 230–239. https://digitalcommons.usu.edu/cgi/viewcontent.cgi?article=1034&context=hwi

Cavallero, L., & Raffaele, E. (2010). Fire enhances the 'competition-free' space of an invader shrub: *Rosa rubiginosa* in northwestern Patagonia. *Biological Invasions, 12*(10), 3395–3404. https://doi.org/10.1007/s10530-010-9738-3

CBD. (2014). *Pathways of introduction of invasive species, their prioritization and management.* 18. https://www.cbd.int/doc/meetings/sbstta/sbstta-18/official/sbstta-18-09-add1-en.pdf

Censky, E., Hodge, K., & Dudley, J. (1998). Over-water dispersal of lizards due to hurricanes. *Nature, 395*, 556. https://doi.org/doi.org/10.1038/26886

Cerri, J., Ciappelli, A., Lenuzza, A., Zaccaroni, M., & Nocita, A. (2018). Recreational angling as a vector of freshwater invasions in Central Italy: Perceptions and prevalence of illegal fish restocking. *Knowledge and Management of Aquatic Ecosystems, 419*. https://doi.org/10.1051/kmae/2018028

Cerri, J., Ferretti, M., & Tricarico, E. (2016). Are you aware of what you are doing? Asking Italian hunters about an invasive alien species they are introducing. *Journal for Nature Conservation, 34*, 15–23. https://doi.org/10.1016/j.jnc.2016.08.003

Chambers, J. C., Bradley, B. A., Brown, C. S., D'Antonio, C., Germino, M. J., Grace, J. B., Hardegree, S. P., Miller, R. F., & Pyke, D. A. (2014). Resilience to Stress and Disturbance, and Resistance to *Bromus tectorum* L. Invasion in Cold Desert Shrublands of Western North America. *Ecosystems, 17*(2), 360–375. https://doi.org/10.1007/s10021-013-9725-5

Chambers, J. C., Roundy, B. A., Blank, R. R., Meyer, S. E., & Whittaker, A. (2007). What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? *Ecological Monographs. 77*(1): 117-145. https://doi.org/10.1890/05-1991

Champagnon, J., Elmberg, J., Guillemain, M., Gauthier-Clerc, M., & Lebreton, J.-D. (2012). Conspecifics can be aliens too: A review of effects of restocking practices in vertebrates. Journal for Nature Conservation, 20(4), 231–241. https://doi.org/10.1016/j.jnc.2012.02.002

Champagnon, J., Legagneux, P., Souchay, G., Inchausti, P., Bretagnolle, V., Bourguemestre, F., Van Ingen, L., & Guillemain, M. (2016). Robust estimation of survival and contribution of captive-bred Mallards *Anas platyrhynchos* to a wild population in a large-scale release programme. *Ibis, 158*(2), 343–352. https://doi.org/10.1111/ibi.12341

Chan, F. T., Stanislawczyk, K., Sneekes, A. C., Dvoretsky, A., Gollasch, S., Minchin, D., David, M., Jelmert, A., Albretsen, J., & Bailey, S. A. (2019). Climate change opens new frontiers for marine species in the Arctic: Current trends and future invasion risks. *Global Change Biology, 25*(1), 25–38. https://doi.org/10.1111/gcb.14469

Chancel, L., Piketty, T., Saez, E., & Zucman, G. (2022). *World Inequality Report 2022*. https://wid.world/news-article/world-inequality-report-2022/

Chandrasekaran, S., Nagendran, N. A., Pandiaraja, D., Krishnankutty, N., & Kamalakannan, B. (2008). Bioinvasion of *Kappaphycus alvarezii* on corals in the Gulf of Mannar, India. *Current Science, 94*(9), 1167–1172. https://www.jstor.org/stable/24100697

Chandrasekaran, S., & Swamy, P. S. (2016). Ecological and Socioeconomic Impacts of *Prosopis juliflora* Invasion in the Semiarid Ecosystems in Selected Villages of Ramnad District in Tamil Nadu. In N. Ghosh, P. Mukhopadhyay, A. Shah, & M. Panda (Eds.), *Nature, Economy and Society: Understanding the Linkages* (pp. 347–357). Springer, India. https://doi.org/10.1007/978-81-322-2404-4_18

Charbonneau, B. R., Wootton, L. S., Wnek, J. P., Langley, J. A., & Posner, M. A. (2017). A species effect on storm erosion: Invasive sedge stabilized dunes more than native grass during Hurricane Sandy. *Journal of Applied Ecology, 54*(5), 1385–1394. https://doi.org/10.1111/1365-2664.12846

Chen, Z., Guo, Y., Yang, J., Ge, J., Li, L., & Chen, G. (2021). Frugivorous birds disperse seeds of *Ligustrum lucidum*, seed-feeding weevils, and parasitic wasps of weevils *via* endozoochory. *Integrative Zoology*. https://doi.org/10.1111/1749-4877.12606

Chinchio, E., Crotta, M., Romeo, C., Drewe, J. A., Guitian, J., & Ferrari, N. (2020). Invasive alien species and disease risk: An open challenge in public and animal health. *PLoS Pathogens, 16*(10), e1008922. https://doi.org/10.1371/journal.ppat.1008922

Chown, S. L., Huiskes, A. H. L., Gremmen, N. J. M., Lee, J. E., Terauds, A., Crosbie, K., Frenot, Y., Hughes, K. A., Imura, S., Kiefer, K., Lebouvier, M., Raymond, B., Tsujimoto, M., Ware, C., Van de Vijver, B., & Bergstrom, D. M. (2012). Continent-wide risk assessment for the establishment of nonindigenous species in Antarctica. *Proceedings of the National Academy of Sciences of the United States of America, 109*(13), 4938–4943. https://doi.org/10.1073/pnas.1119787109

Chucholl, C. (2013). Invaders for sale: Trade and determinants of introduction of ornamental freshwater crayfish. *Biological Invasions, 15*(1), 125–141. https://doi.org/10.1007/s10530-012-0273-2

Chwedorzewska, K. J. (2008). *Poa annua* L. in Antarctic: Searching for the source of introduction. *Polar Biology, 31*(3), Article 3. https://doi.org/10.1007/s00300-007-0353-4

Chwedorzewska, K. J., Gielwanowska, I., Olech, M., Molina-Montenegro, M. A., Wodkiewicz, M., & Galera, H. (2015). *Poa annua* L. in the maritime Antarctic: An overview. *POLAR RECORD, 51*(6), 637–643. https://doi.org/10.1017/S0032247414000916

Chwedorzewska, K. J., Korczak-Abshire, M., & Znój, A. (2020). Is Antarctica under threat of alien species invasion? *Global Change Biology, 26*(4), 1942–1943. https://doi.org/10.1111/gcb.15013

Clarke, H., & Evans, J. P. (2019). Exploring the future change space for fire weather in southeast Australia. *Theoretical and Applied Climatology, 136*(1), 513–527. https://doi.org/10.1007/s00704-018-2507-4

Clavero, M. (2016). Species substitutions driven by anthropogenic positive feedbacks: Spanish crayfish species as a case study. *Biological Conservation, 193*, 80–85. https://doi.org/10.1016/j.biocon.2015.11.017

Clergeau, P., & Yésou, P. (2006). Behavioural Flexibility and Numerous Potential Sources of Introduction for the Sacred Ibis: Causes of Concern in Western Europe? *Biological Invasions, 8*(6), 1381–1388. https://doi.org/10.1007/s10530-006-0002-9

Clout, M. N. (2006). Keystone aliens? The multiple impacts of brushtail possums. In R. B. Allen & W. G. Lee (Eds.), *Biological Invasions in New Zealand* (pp. 265–279). Springer. https://link.springer.com/chapter/10.1007/3-540-30023-6_17

Cofrancesco Jr, A. F., Reaves, D. R., & Averett, D. E. (2007). *Transfer of invasive species associated with the movement of military equipment and personnel.* Engineer Research and Development Center MS Environmental lab. https://apps.dtic.mil/sti/pdfs/ADA470914.pdf

Cohen, J., Mirotchnick, N., & Leung, B. (2007). Thousands introduced annually: The aquarium pathway for non-indigenous plants to the St Lawrence Seaway. *Frontiers in Ecology and the Environment, 5*(10), 528–532. https://doi.org/10.1890/060137

Colautti, R. I., & MacIsaac, H. J. (2004). A neutral terminology to define 'invasive' species. *Diversity and Distributions, 10*(2), Article 2. https://doi.org/10.1111/j.1366-9516.2004.00061.x

Coldsnow, K. D., & Relyea, R. A. (2018). Toxicity of various road-deicing salts to Asian clams *(Corbicula fluminea)*. *Environmental Toxicology and Chemistry, 37*(7), 1839–1845. https://doi.org/10.1002/etc.4126

Coles, S. L., DeFelice, R. C., Eldredge, L. G., & Carlton, J. T. (1999). Historical and recent introductions of non-indigenous marine species into Pearl Harbor, Oahu, Hawaiian Islands. *Marine Biology, 135*(1), 147–158. https://doi.org/10.1007/s002270050612

Conn, J. E., Wilkerson, R. C., Segura, M. N. O., de Souza, R. T. L., Schlichting, C. D., Wirtz, R. A., & Póvoa, M. M. (2002). Emergence of a new neotropical malaria vector facilitated by human migration and changes in land use. *The American Journal of Tropical Medicine and Hygiene, 66*(1), 18–22. https://doi.org/10.4269/ajtmh.2002.66.18

Constantinescu, C., Mattoo, A., & Ruta, M. (2016). Does the global trade slowdown matter? *Journal of Policy Modeling, 38*(4), 711–722. https://doi.org/10.1016/j.jpolmod.2016.05.013

Cook, G. D., & Dias, L. (2006). It was no accident: Deliberate plant introductions by Australian government agencies during the 20th century. *Australian Journal of Botany, 54*(7), 601–625. https://doi.org/10.1071/BT05157

Copp, G. H., Vilizzi, L., & Gozlan, R. E. (2010). The demography of introduction pathways, propagule pressure and occurrences of non-native freshwater fish in England. *Aquatic Conservation: Marine and Freshwater Ecosystems, 20*(5), 595–601. https://doi.org/10.1002/aqc.1129

Cote, I. M., Darling, E. S., & Brown, C. J. (2016). Interactions among ecosystem stressors and their importance in conservation. *Proceedings of the Royal Society B-Biological Sciences, 283*(20152592). https://doi.org/10.1098/rspb.2015.2592

Cowie, R. H. (2001). Can snails ever be effective and safe biocontrol agents? *International Journal of Pest Management, 47*(1), 23–40. https://doi.org/10.1080/09670870150215577

Cowling, J. E., Spicer, J. I., Weeks, J. M., & Gaston, K. J. (2003). Environmental tolerances of an invasive terrestrial amphipod, *Arcitalitrus dorrieni* (Hunt) in Britain. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology, 136*(3), 735–747. https://doi.org/10.1016/S1095-6433(03)00242-3

Cox, G. W. (1999). *Alien Species in North America and Hawaii*. Island Press. https://islandpress.org/books/alien-species-north-america-and-hawaii

Crane, E. (1999). *The World History of Beekeeping and Honey Hunting.* Routledge. https://www.routledge.com/The-World-History-of-Beekeeping-and-Honey-Hunting/Crane/p/book/9780415924672

Cusack, D. F., Karpman, J., Ashdown, D., Cao, Q., Ciochina, M., Halterman, S., Lydon, S., & Neupane, A. (2016). Global change effects on humid tropical forests: Evidence for biogeochemical and biodiversity shifts at an ecosystem scale. *Reviews of Geophysics, 54*(3), Article 3. https://doi.org/10.1002/2015RG000510

da Rosa, C. A., de Almeida Curi, N. H., Puertas, F., & Passamani, M. (2017). Alien terrestrial mammals in Brazil: Current status and management. *Biological Invasions, 19*(7), 2101–2123. https://doi.org/10.1007/s10530-017-1423-3

Daehler, C. C. (2003). Performance Comparisons of Co-Occurring Native and Alien Invasive Plants: Implications for Conservation and Restoration. *Annual Review of Ecology, Evolution, and Systematics, 34*(1), 183–211. https://doi.org/10.1146/annurev.ecolsys.34.011802.132403

Dafforn, K. A., Glasby, T. M., Airoldi, L., Rivero, N. K., Mayer-Pinto, M., & Johnston, E. L. (2015). Marine urbanization: An ecological framework for designing multifunctional artificial structures. *Frontiers in Ecology and the Environment, 13*(2), 82–90. https://doi.org/10.1890/140050

Dainese, M., Aikio, S., Hulme, P. E., Bertolli, A., Prosser, F., & Marini, L. (2017). Human disturbance and upward expansion of plants in a warming climate. *Nature Climate Change, 7*(8), 577–580. https://doi.org/10.1038/nclimate3337

Dainese, M., Kühn, I., & Bragazza, L. (2014). Alien plant species distribution in the European Alps: Influence of species' climatic requirements. *Biological Invasions, 16*(4), 815–831. https://doi.org/10.1007/s10530-013-0540-x

Dalsimer, A. (2002). Why DOD is interested in invasive species. *Federal Facilities Environmental Journal, 13*(3), 41–54. https://doi.org/10.1002/ffej.10044

D'Amore, D. M., Popescu, V. D., & Morris, M. R. (2019). The influence of the invasive process on behaviours in an intentionally introduced hybrid, *Xiphophorus helleri–maculatus. Animal Behaviour, 156*, 79–85. https://doi.org/10.1016/j.anbehav.2019.08.009

D'Antonio, C. M. (2000). Fire, plant invasions, and global changes. In H. A. Mooney & R. J. Hobbs (Eds.), *Invasive species in a changing world* (pp. 65–93). Island PRess.

D'Antonio, C. M., & Meyerson, L. A. (2002). Exotic plant species as problems and solutions in ecological restoration: A synthesis. *Restoration Ecology, 10*(4), 703–713. https://doi.org/10.1046/j.1526-100X.2002.01051.x

D'Antonio, C. M., & Vitousek, P. M. (1992). Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change. *Annual Review of Ecology and Systematics, 23*, 63–87. JSTOR. https://doi.org/annurev.es.23.110192.000431

Darwin, C. (1859). *On the origin of species by means of natural selection, or, The preservation of favoured races in the struggle for life*. John Murray.

Daskalov, G. M., Grishin, A. N., Rodionov, S., & Mihneva, V. (2007). Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. *Proceedings of the National Academy of Sciences of the United States of America, 104*(25), 10518–10523. https://doi.org/10.1073/pnas.0701100104

Davidson, J. M., Wickland, A. C., Patterson, H. A., Falk, K. R., & Rizzo, D. M. (2005). Transmission of *Phytophthora ramorum* in mixed-evergreen forest in California. *Phytopathology, 95*(5), 587–596. https://doi.org/10.1094/PHYTO-95-0587

Davis, A. J. S., & Darling, J. A. (2017). Recreational freshwater fishing drives non-native aquatic species richness patterns at a continental scale. *Diversity and Distributions, 23*(6), 692–702. https://doi.org/10.1111/ddi.12557

Davis, K. T., Maxwell, B. D., Caplat, P., Pauchard, A., & Nuñez, M. A. (2019). Simulation model suggests that fire promotes lodgepole pine *(Pinus contorta)* invasion in Patagonia. *Biological Invasions, 21*(7), 2287–2300. https://doi.org/10.1007/s10530-019-01975-1

Davis, M. A., Grime, J. P., & Thompson, K. (2000). Fluctuating resources in plant communities: A general theory of invasibility. *Journal of Ecology, 88*(3), 528–534. https://doi.org/10.1046/j.1365-2745.2000.00473.x

Davis, N. E., Bennett, A., Forsyth, D. M., Bowman, D. M. J. S., Lefroy, E. C., Wood, S. W., Woolnough, A. P., West, P., Hampton, J. O., Johnson, C. N., Davis, N. E., Bennett, A., Forsyth, D. M., Bowman, D. M. J. S., Lefroy, E. C., Wood, S. W., Woolnough, A. P., West, P., Hampton, J. O., & Johnson, C. N. (2016). A systematic review of the impacts and management of introduced deer (family Cervidae) in Australia. *Wildlife Research, 43*(6), Article 6. https://doi.org/10.1071/WR16148

Davis, N. E., Forsyth, D. M., & Coulson, G. (2010). Facilitative interactions between an exotic mammal and native and exotic plants: Hog deer *(Axis porcinus)* as seed dispersers in south-eastern Australia. *Biological Invasions, 12*(5), 1079–1092. https://doi.org/10.1007/s10530-009-9525-1

Dawson, W., Moser, D., van Kleunen, M., Kreft, H., Pergl, J., Pyšek, P., Weigelt, P., Winter, M., Lenzner, B., Blackburn, T. M., Dyer, E. E., Cassey, P., Scrivens, S. L., Economo, E. P., Guénard, B., Capinha, C., Seebens, H., García-Díaz, P., Nentwig, W., … Essl, F. (2017). Global hotspots and correlates of alien species richness across taxonomic groups. *Nature Ecology & Evolution, 1*(7), 0186. https://doi.org/10.1038/s41559-017-0186

Day, M. D., Cock, M. J. W., Conant, P., Cooke, B., Furlong, M. J., Paynter, Q., Ramadan, M. M., & Wright, M. G. (2021). Biological control successes and failures: Oceania region. In P. G. Mason (Ed.), *Biological control: Global impacts, challenges and future directions of pest management* (pp. 334–367). CSIRO. https://ebooks.publish.csiro.au/content/biological-control

Day, M. J. (2011). One health: The importance of companion animal vector-borne diseases. *Parasites & Vectors, 4*(1), 49. https://doi.org/10.1186/1756-3305-4-49

De La Riva, D. G., & Trumble, J. T. (2016). Selenium exposure results in reduced reproduction in an invasive ant species and altered competitive behavior for a native ant species. *Environmental Pollution, 213*, 888–894. https://doi.org/10.1016/j.envpol.2016.03.045

De Silva, S. S. (2012). Aquaculture: A newly emergent food production sector—and perspectives of its impacts on biodiversity and conservation. *Biodiversity and Conservation, 21*(12), 3187–3220. https://doi.org/10.1007/s10531-012-0360-9

De Silva, S. S., Nguyen, T. T. T., Abery, N. W., & Amarasinghe, U. S. (2006). An evaluation of the role and impacts of alien finfish in Asian inland aquaculture. *Aquaculture Research, 37*(1), 1–17. https://doi.org/10.1111/j.1365-2109.2005.01369.x

De Silva, S. S., Nguyen, T. T. T., Turchini, G. M., Amarasinghe, U. S., & Abery, N. W. (2009). Alien Species in Aquaculture and Biodiversity: A Paradox in Food Production. *AMBIO: A Journal of the Human Environment, 38*(1), 24–28. https://doi.org/10.1579/0044-7447-38.1.24

de Villalobos, A. E., & Schwerdt, L. (2020). Seasonality of feral horse grazing and invasion of *Pinus halepensis* in grasslands of the Austral Pampean Mountains (Argentina): Management considerations. *Biological Invasions, 22*(10), 2941–2955. https://doi.org/10.1007/s10530-020-02300-x

de Villalobos, A. E., & Zalba, S. M. (2010). Continuous feral horse grazing and grazing exclusion in mountain pampean grasslands in Argentina. *Acta Oecologica, 36*(5), 514–519. https://doi.org/10.1016/j.actao.2010.07.004

Dehnen-Schmutz, K., Touza, J., Perrings, C., & Williamson, M. (2007). A century of the ornamental plant trade and its impact on invasion success. *Diversity and Distributions, 13*(5), Article 5. https://doi.org/10.1111/j.1472-4642.2007.00359.x

Delibes-Mateos, M. (2015). Game managers' views on the release of farm-reared red-legged partridges in hunting estates within central Spain. Journal for *Nature Conservation, 26*, 1–8.

Delibes-Mateos, M., Giergiczny, M., Caro, J., Viñuela, J., Riera, P., & Arroyo, B. (2014). Does hunters' willingness to pay match the best hunting options for biodiversity conservation? A choice experiment application for small-game hunting in Spain. *Biological Conservation, 177*, 36–42. https://doi.org/10.1016/j.biocon.2014.06.004

Denny, E., Yakovlevich, P., Eldridge, M. D. B., & Dickman, C. (2002). Social and genetic analysis of a population of free-living cats (Felis catus L.) exploiting a resource-rich habitat. *Wildlife Research, 29*(4), 405. https://doi.org/10.1071/WR02092

Devin, S., Bollache, L., Noël, P.-Y., & Beisel, J.-N. (2005). Patterns of Biological Invasions in French Freshwater Systems by Non-Indigenous Macroinvertebrates. *Hydrobiologia, 551*(1), 137–146. https://doi.org/10.1007/s10750-005-4456-z

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J. R., Arico, S., Báldi, A., Bartuska, A., Baste, I. A., Bilgin, A., Brondizio, E., Chan, K. M., Figueroa, V. E., Duraiappah, A., Fischer, M., Hill, R., … Zlatanova, D. (2015). The IPBES Conceptual Framework—Connecting nature and people. *Current Opinion in Environmental Sustainability, 14*, 1–16. https://doi.org/10.1016/j.cosust.2014.11.002

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., Hill, R., Chan, K. M. A., Baste, I. A., Brauman, K. A., Polasky, S., Church, A., Lonsdale, M., Larigauderie, A., Leadley, P. W., van Oudenhoven, A. P. E., van der Plaat, F., Schröter, M., Lavorel, S., … Shirayama, Y. (2018). An inclusive approach to assesse nature's contributions to people. *Science, 359*(6373), 270–272. https://doi.org/10.1126/science.aap8826

Diaz Velez, M. C., Ferreras, A. E., & Paiaro, V. (2020). Seed dispersal interactions promoting plant invasions. In *Plant Invasions: The role of bitic interactions.* CABI International. https://ri.conicet.gov.ar/handle/11336/138181

Dickie, I. A., Bennett, B. M., Burrows, L. E., Nuñez, M. A., Peltzer, D. A., Porté, A., Richardson, D. M., Rejmánek, M., Rundel, P. W., & van Wilgen, B. W. (2014). Conflicting values: Ecosystem services and invasive tree management. *Biological Invasions, 16*(3), 705–719. https://doi.org/10.1007/s10530-013-0609-6

Dickie, I. A., Bolstridge, N., Cooper, J. A., & Peltzer, D. A. (2010). Co-invasion by *Pinus* and its mycorrhizal fungi. *New Phytologist, 187*(2), 475–484. https://doi.org/10.1111/j.1469-8137.2010.03277.x

Diez, J. M., D'Antonio, C. M., Dukes, J. S., Grosholz, E. D., Olden, J. D., Sorte, C. J., Blumenthal, D. M., Bradley, B. A., Early, R., Ibáñez, I., Jones, S. J., Lawler, J. J., & Miller, L. P. (2012). Will extreme climatic events facilitate biological invasions? *Frontiers in Ecology and the Environment, 10*(5), 249–257. https://doi.org/10.1890/110137

Dodd, A. J., Burgman, M. A., McCarthy, M. A., & Ainsworth, N. (2015). The changing patterns of plant naturalization in Australia. *Diversity and Distributions, 21*(9), Article 9. https://doi.org/10.1111/ddi.12351

Donaldson, J. E., Hui, C., Richardson, D. M., Robertson, M. P., Webber, B. L., & Wilson, J. R. U. (2014). Invasion trajectory of alien trees: The role of introduction pathway and planting history. *Global Change Biology, 20*(5), 1527–1537. https://doi.org/10.1111/gcb.12486

Doney, S. C., Ruckelshaus, M., Duffy, J. E., Barry, J. P., Chan, F., English, C. A., Galindo, H. M., Grebmeier, J. M., Hollowed, A. B., Knowlton, N., Polovina, J., Rabalais, N. N., Sydeman, W. J., & Talley, L. D. (2012). Climate Change Impacts on Marine Ecosystems. *Annual Review of Marine Science, Vol 4, 4*, 11–37.

Dostál, P., Müllerová, J., Pyšek, P., Pergl, J., & Klinerová, T. (2013). The impact of an invasive plant changes over time. *Ecology Letters, 16*(10), 1277–1284. https://doi.org/10.1111/ele.12166

Downing, J., Borrero, H., & Liu, H. (2016). Differential impacts from an extreme cold spell on subtropical vs. Tropical specialist bees in southern Florida. *Ecosphere, 7*(5), e01302. https://doi.org/10.1002/ecs2.1302

Driscoll, D. A., Catford, J. A., Barney, J. N., Hulme, P. E., Inderjit, Martin, T. G., Pauchard, A., Pyšek, P., Richardson, D. M., Riley, S., & Visser, V. (2014). New pasture plants intensify invasive species risk. *Proceedings of the National Academy of Sciences, 111*(46), 16622–16627. https://doi.org/10.1073/pnas.1409347111

Duenn, P., Salpeteur, M., & Reyes-García, V. (2017). Rabari Shepherds and the Mad Tree: The Dynamics of Local Ecological Knowledge in the Context of *Prosopis juliflora* Invasion in Gujarat, India. *Journal of Ethnobiology, 37*(3), 561–580. https://doi.org/10.2993/0278-0771-37.3.561

Duffy, D. J., & Lepczyk, C. A. (2021). The Historical Ecology of Game Species Introductions in Hawai'i1. *Pacific Science, 75*(1). https://doi.org/10.2984/75.1.1

Duffy, G. A., Coetzee, B. W. T., Latombe, G., Akerman, A. H., McGeoch, M. A., & Chown, S. L. (2017). Barriers to globally invasive species are weakening across the Antarctic. *Diversity and Distributions, 23*(9), 982–996. https://doi.org/10.1111/ddi.12593

Dukes, J. S. (2000). Will the increasing atmospheric CO_2 concentration affect the success of invasive species? In *Invasive species in a changing world* (Vol. 95). Island Press.

Dukes, J. S., Pontius, J., Orwig, D., Garnas, J. R., Rodgers, V. L., Brazee, N., Cooke, B., Theoharides, K. A., Stange, E. E., Harrington, R., Ehrenfeld, J., Gurevitch, J., Lerdau, M., Stinson, K., Wick, R., & Ayres, M. (2009). Responses of insect pests, pathogens, and invasive plant species to climate change in the forests of northeastern North America: What can we predict? *Canadian Journal of Forest Research, 39*(2), 231–248. https://doi.org/10.1139/X08-171

Dulac, J. (2013). *Global land transport infrastructure requirements. Estimating road and railway infrastructure capacity and costs to 2050*. International Energy Agency. https://www.ctc-n.org/sites/www.ctc-n.org/files/resources/transportinfrastructureinsights_final_web.pdf

Dunham, J. B., Young, M. K., Gresswell, R. E., & Rieman, B. E. (2003). Effects of fire on fish populations: Landscape perspectives on persistence of native fishes and nonnative fish invasions. *Forest Ecology and Management, 178*(1), 183–196. https://doi.org/10.1016/S0378-1127(03)00061-6

Dyer, E. E., Cassey, P., Redding, D. W., Collen, B., Franks, V., Gaston, K. J., Jones, K. E., Kark, S., Orme, C. D. L., & Blackburn, T. M. (2017). The global distribution and drivers of alien bird species richness. *PLoS Biology, 15*(1), e2000942. https://doi.org/10.1371/journal.pbio.2000942

Early, R., Bradley, B. A., Dukes, J. S., Lawler, J. J., Olden, J. D., Blumenthal, D. M., Gonzalez, P., Grosholz, E. D., Ibañez, I., Miller, L. P., Sorte, C. J. B., & Tatem, A. J. (2016). Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications, 7*(1), 12485. https://doi.org/10.1038/ncomms12485

Echandi, E., Knoke, J. K., Nigh, E. L., Shenk, M., & Weekman, G. T. (1972). *Crop Protection in Brazil, Uruguay, Bolivia, Ecuador, and Dominican Republic.* United States Agency for International Development, Washington, DC. https://agris.fao.org/agris-search/search.do?recordID=XF2016078207

Ehrenfeld, J. G. (2008). Exotic invasive species in urban wetlands: Environmental correlates and implications for wetland management. *Journal of Applied Ecology, 45*(4), 1160–1169. https://doi.org/10.1111/j.1365-2664.2008.01476.x

Ellender, B. R., & Weyl, O. L. F. (2014). A review of current knowledge, risk and ecological impacts associated with non-native freshwater fish introductions in South Africa. *Aquatic Invasions, 9*(2), 117–132. https://doi.org/10.3391/ai.2014.9.2.01

Eller, F., Lambertini, C., Nguyen, L. X., & Brix, H. (2014). Increased Invasive Potential of Non-Native *Phragmites australis*: Elevated CO_2 and Temperature Alleviate Salinity Effects on Photosynthesis and Growth. *Global Change Biology, 20*(2), 531–543. https://doi.org/10.1111/gcb.12346

Ellstrand, N. C., Meirmans, P., Rong, J., Bartsch, D., Ghosh, A., de Jong, T. J., Haccou, P., Lu, B.-R., Snow, A. A., Neal Stewart, C., Strasburg, J. L., van Tienderen, P. H., Vrieling, K., & Hooftman, D. (2013). Introgression of Crop Alleles into Wild or Weedy Populations. *Annual Review of Ecology, Evolution, and Systematics, 44*(1), 325–345. https://doi.org/10.1146/annurev-ecolsys-110512-135840

Elorza, M. S., Sánchez, E. D. D., & Vesperinas, E. S. (2004). *Atlas de las plantas alóctonas invasoras en España.* Dirección General de la Biodiversidad. Ministerio de Medio Ambiente.

Elton, C. S. (1958). *The Ecology of Invasions by Animals and Plants.* University of Chicago Press. http://link.springer.com/10.1007/978-1-4899-7214-9

Englund, R. A. (2002). The loss of native biodiversity and continuing nonindigenous species introductions in freshwater, estuarine, and wetland communities of Pearl Harbor, Oahu, Hawaiian Islands. *Estuaries, 25*(3), 418–430. https://doi.org/10.1007/BF02695984

Ennos, R., Cottrell, J., Hall, J., & O'Brien, D. (2019). Is the introduction of novel exotic forest tree species a rational response to rapid environmental change? – A British perspective. *Forest Ecology and Management, 432*, 718–728. https://doi.org/10.1016/j.foreco.2018.10.018

EPPO. (1996). *Situation of Diabrotica virgifera in Serbia (YU). From International Workshop "Western Corn Rootworm in Europe 95"1995-11-08.* (1996/006; EPPO Reporting Service).

Eriksen, M., Lebreton, L. C., Carson, H. S., Thiel, M., Moore, C. J., Borerro, J. C., Galgani, F., Ryan, P. G., & Reisser, J. (2014). Plastic pollution in the world's oceans: More than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. *PLoS ONE, 9*(12), e111913. https://doi.org/10.1371/ journal.pone.0111913

Eriksson, O., Cousins, S. A. O., & Bruun, H. H. (2002). Land-use history and fragmentation of traditionally managed grasslands in Scandinavia. *Journal of Vegetation Science, 13*(5), 743–748. https://doi.org/10.1111/j.1654-1103.2002.tb02102.x

Essink, K. (2003). Response of an estuarine ecosystem to reduced organic waste discharge. *Aquatic Ecology, 37*(1), 65–76. https://doi.org/10.1023/A:1022139600926

Essl, F., Dawson, W., Kreft, H., Pergl, J., Pyšek, P., Van Kleunen, M., Weigelt, P., Mang, T., Dullinger, S., Lenzner, B., Moser, D., Maurel, N., Seebens, H., Stein, A., Weber, E., Chatelain, C., Inderjit, Genovesi, P., Kartesz, J., … Winter, M. (2019). Drivers of the relative richness of naturalized and invasive plant species on Earth. *AoB PLANTS, 11*(5), plz051. https://doi.org/10.1093/aobpla/plz051

Essl, F., Dullinger, S., Moser, D., Steinbauer, K., & Mang, T. (2015). Macroecology of global bryophyte invasions at different invasion stages. *Ecography, 38*(5), 488–498. https://doi.org/10.1111/ecog.00905

Essl, F., Dullinger, S., Rabitsch, W., Hulme, P. E., Hülber, K., Jarošík, V., Kleinbauer, I., Krausmann, F., Kühn, I., Nentwig, W., Vilà, M., Genovesi, P., Gherardi, F., Desprez-Loustau, M.-L., Roques, A., & Pyšek, P. (2011). Socioeconomic legacy yields an invasion debt. *Proceedings of the National Academy of Sciences, 108*(1), 203–207. https://doi.org/10.1073/pnas.1011728108

Estévez, R. A., Anderson, C. B., Pizarro, J. C., & Burgman, M. A. (2015). Clarifying values, risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species management. *Conservation Biology, 29*(1), 19–30. https://doi.org/10.1111/cobi.12359

Evans, E. W., Soares, A. O., & Yasuda, H. (2011). Invasions by ladybugs, ladybirds, and other predatory beetles. *BioControl, 56*(4), 597–611. https://doi.org/10.1007/s10526-011-9374-6

Evans, T., Ermgassen, P. zu, Amano, T., & Peh, K. S.-H. (2018). Does governance play a role in the distribution of invasive alien species? *Ecology and Evolution, 8*(4), 1984–1994. https://doi.org/10.1002/ece3.3744

Eversmeyer, M. G., & Kramer, C. L. (2000). Epidemiology of wheat leaf and stem rust in the central great plains of the USA. *Annual Review of Phytopathology, 38*, 491. https://doi.org/10.1146/annurev.phyto.38.1.491

Fabry, V., McClintock, J., Mathis, J., & Grebmeier, J. (2009). Ocean Acidification at High Latitudes: The Bellwether. *Oceanography, 22*(4), 160–171. https://doi.org/10.5670/oceanog.2009.105

Fagúndez, J. (2013). Heathlands confronting global change: Drivers of biodiversity loss from past to future scenarios. *Annals of Botany, 111*(2), Article 2. https://doi.org/10.1093/aob/mcs257

FAO. (1995). *Guidelines for pest risk analysis.* International standards for phytosanitary measures (ISPM). No. 2. Food and Agriculture Organization of the United Nations. https://www.ippc.int/largefiles/adopted_ISPMs_previousversions/en/ISPM_02_1995_En_2006-05-03.pdf

FAO. (2001). *The State of Food and Agriculture 2001* (Annual Report SBN 92-5-104600-X). Food and Agriculture Organization of the United Nations. http://www.fao.org/3/x9800e/x9800e00.htm

FAO. (2015). *Status of the World's Soil Resources: Main Report.* FAO. http://www.fao.org/documents/card/fr/c/c6814873-efc3-41db-b7d3-2081a10ede50/

FAO. (2021). *Towards Blue Transformation*. www.fao.org. https://doi.org/10.4060/cc0461en

Farr, H., Ruttenberg, B., Walter, R. K., Wang, Y.-H., & White, C. (2021). Potential environmental effects of deepwater floating offshore wind energy facilities. *Ocean & Coastal Management, 207*, 105611. https://doi.org/10.1016/j.ocecoaman.2021.105611

Faulkes, Z. (2010). The spread of the parthenogenetic marbled crayfish, Marmorkrebs (*Procambarus* sp.), in the North American pet trade. *Aquatic Invasions, 5*(4), 447–450. https://doi.org/10.3391/ai.2010.5.4.16

Faulkes, Z. (2015). The global trade in crayfish as pets. *Crustacean Research, 44*(0), 75–92. https://doi.org/10.18353/crustacea.44.0_75

Faulkner, K. T., Burness, A., Byrne, M. J., Kumschick, S., Peters, K., Robertson, M. P., Saccaggi, D. L., Weyl, O. L. F., & Williams, V. L. (2020). South Africa's pathways of introduction and dispersal and how they have changed over time. In B. W. van Wilgen, J. Measey, D. W. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological invasions in South Africa* (Vol. 14, pp. 311–352). Springer. https://doi.org/10.1007/978-3-030-32394-3_12

Faulkner, K. T., Robertson, M. P., Rouget, M., & Wilson, J. R. U. (2016). Understanding and managing the introduction pathways of alien taxa: South Africa as a case study. *Biological Invasions, 18*(1), Article 1. https://doi.org/10.1007/s10530-015-0990-4

Faust, E., Halvorsen, K. T., Andersen, P., Knutsen, H., & André, C. (2018). Cleaner fish escape salmon farms and hybridize with local wrasse populations. *Royal Society Open Science, 5*(3), 171752. https://doi.org/10.1098/rsos.171752

Febbraro, M. D., Menchetti, M., Russo, D., Ancillotto, L., Aloise, G., Roscioni, F., Preatoni, D. G., Loy, A., Martinoli, A., Bertolino, S., & Mori, E. (2019). Integrating climate and land-use change scenarios in modelling the future spread of invasive squirrels in Italy. *Diversity and Distributions, 25*(4), 644–659. https://doi.org/10.1111/ddi.12890

Feehan, C., Scheibling, R. E., Brown, M. S., & Thompson, K. R. (2016). Marine epizootics linked to storms: Mechanisms of pathogen introduction and persistence inferred from coupled physical and biological time-series. *Limnology and Oceanography, 61*, 316–329. https://doi.org/10.1002/lno.10217

Feng, Y., Fouqueray, T. D., & van Kleunen, M. (2019). Linking Darwin's naturalisation hypothesis and Elton's diversity-invasibility hypothesis in experimental grassland communities. *Journal of Ecology, 107*(2), 794–805. https://doi.org/10.1111/1365-2745.13061

Fernandes, P., Antunes, C., Pinho, P., Máguas, C., & Correia, O. (2016). Natural regeneration of *Pinus pinaster* and *Eucalyptus globulus* from plantation into adjacent natural habitats. *Forest Ecology and Management, 378*, 91–102. https://doi.org/10.1016/j.foreco.2016.07.027

Fèvre, E. M., Bronsvoort, B. M. de C., Hamilton, K. A., & Cleaveland, S. (2006). Animal movements and the spread of infectious diseases. *Trends in Microbiology, 14*(3), 125–131. https://doi.org/10.1016/j.tim.2006.01.004

Fisher, M. C., & Garner, T. W. J. (2007). The relationship between the emergence of *Batrachochytrium dendrobatidis*, the international trade in amphibians and introduced amphibian species. *Fungal Biology Reviews, 21*(1), 2–9. https://doi.org/10.1016/j.fbr.2007.02.002

Fisher, M. C., & Garner, T. W. J. (2020). Chytrid fungi and global amphibian declines. *Nature Reviews Microbiology, 18*(6), 332–343. https://doi.org/10.1038/s41579-020-0335-x

Floc'h, J. Y., Pajot, R., & Wallentinus, I. (1991). The Japanese brown alga *Undaria pinnatifida* on the coast of France and its possible establishment in European waters. *ICES Journal of Marine Science, 47*(3), 379–390. https://doi.org/10.1093/icesjms/47.3.379

Fløjgaard, C., Morueta-Holme, N., Skov, F., Madsen, A. B., & Svenning, J.-C. (2009). Potential 21st century changes to the mammal fauna of Denmark – implications of climate change, land-use, and invasive species. *IOP Conference Series: Earth and Environmental Science, 8*, 012016. https://doi.org/10.1088/1755-1315/8/1/012016

Flory, S. L., Lorentz, K. A., Gordon, D. R., & Sollenberger, L. E. (2012). Experimental approaches for evaluating the invasion risk of biofuel crops. *Environmental Research Letters, 7*(4), 045904. https://doi.org/10.1088/1748-9326/7/4/045904

Fobert, E., Fox, M. G., Ridgway, M., & Copp, G. H. (2011). Heated competition: How climate change will affect non-native pumpkinseed *Lepomis gibbosus* and native perch *Perca fluviatilis* interactions in the UK. *Journal of Fish Biology, 79*(6), 1592–1607.

Fonseca, É., Both, C., & Cechin, S. Z. (2019). Introduction pathways and socio-economic variables drive the distribution of alien amphibians and reptiles in a megadiverse country. *Diversity and Distributions, 25*(7), 1130–1141. https://doi.org/10.1111/ddi.12920

Fontúrbel, F. E. (2020). Mistletoes in a changing world: A premonition of a non-analog future? *Botany, 98*(9), 479–488. https://doi.org/10.1139/cjb-2019-0195

Forseth, I. N., & Innis, A. F. (2004). Kudzu *(Pueraria montana)*: History, physiology, and ecology combine to make a major ecosystem threat. *Critical Reviews in Plant Sciences, 23*(5), 401–413. https://doi.org/10.1080/07352680490505150

Forsyth, G. G., Richardson, D. M., Brown, P. J., & van Wilgen, B. W. (2004). A rapid assessment of the invasive status of *Eucalyptus* species in two South African provinces. *South African Journal of Science.* https://open.uct.ac.za/handle/11427/24328

Fosberg F.R. (1957). The naturalised flora of Micronesia and World War II. *Proceedings of the 8th Pacific Science Congress, 4*, 229–234.

Fowler, A. C., Eadie, J. M., & Engilis, A. (2009). Identification of endangered Hawaiian ducks *(Anas wyvilliana)*, introduced North American mallards *(A. platyrhynchos)* and their hybrids using multilocus genotypes. *Conservation Genetics, 10*(6), 1747. https://doi.org/10.1007/s10592-008-9778-8

Fowles, A. E., Edgar, G. J., Stuart-Smith, R. D., Kirkpatrick, J. B., Hill, N., Thomson, R. J., & Strain, E. M. A. (2018). Effects of Pollution From Anthropogenic Point Sources on the Recruitment of Sessile Estuarine Reef Biota. *Frontiers in Marine Science, 5*, 417. https://doi.org/10.3389/fmars.2018.00417

Foxcroft, L. C., Richardson, D. M., & Wilson, J. R. U. (2008). Ornamental Plants as Invasive Aliens: Problems and Solutions in Kruger National Park, South Africa. *Environmental Management, 41*(1), 32–51. https://doi.org/10.1007/s00267-007-9027-9

Franklin, J. A., Zipper, C. E., Burger, J. A., Skousen, J. G., & Jacobs, D. F. (2012). Influence of herbaceous ground cover on forest restoration of eastern US coal surface mines. *New Forests, 43*(5), 905–924. https://doi.org/10.1007/s11056-012-9342-8

Franzese, J., & Raffaele, E. (2017). Fire as a driver of pine invasions in the Southern Hemisphere: A review. *Biological Invasions, 19*(8), Article 8. https://doi.org/10.1007/s10530-017-1435-z

Fraser, C. I., Dutoit, L., Morrison, A. K., Pardo, L. M., Smith, S. D. A., Pearman, W. S., Parvizi, E., Waters, J., & Macaya, E. C. (2022). Southern Hemisphere coasts are biologically connected by frequent, long-distance rafting events. *Current Biology*, *32*(14), 3154-3160.e3. https://doi.org/10.1016/j.cub.2022.05.035

Fraser, C. I., Morrison, A. K., McC Hogg, A., Macaya, E. C., Sebille, E. van, Ryan, P. G., Padovan, A., Jack, C., Validivia, N., & Waters, J. M. (2018). Antarctica's ecological isolation will be broken by storm-driven dispersal and warming. *Nature Climate Change, 8*(8), 704–708. https://doi.org/10.1038/s41558-018-0209-7

Fraser, E. J., Harrington, L. A., Macdonald, D. W., & Lambin, X. (2017). Control of an invasive species: The American mink in Great Britain. In *Biology and Conservation of Musteloids.* Oxford University Press. https://doi.org/10.1093/oso/9780198759805.003.0016

Freifeld, H. B., Plentovich, S., Farmer, C., Kohley, C. R., Luscomb, P., Work, T. M., Tsukayama, D., Wallace, G. E., MacDonald, M. A., & Conant, S. (2016). Long-distance translocations to create a second millerbird population and reduce extinction risk. *Biological Conservation, 199*, 146–156. https://doi.org/10.1016/j.biocon.2016.05.006

Frerichs, R. R., Keim, P. S., Barrais, R., & Piarroux, R. (2012). Nepalese origin of cholera epidemic in Haiti. *Clinical Microbiology and Infection, 18*(6), E158–E163. https://doi.org/10.1111/j.1469-0691.2012.03841.x

Froud-Williams, R. J. (1996). Weeds and climate change: Implications for their ecology and control. *Aspects of Applied Biology (United Kingdom)*. https://agris.fao.org/agris-search/search.do?recordID=GB9629078

Fuentes-Lillo, E., Cuba-Díaz, M., Troncoso-Castro, J. M., & Rondanelli-Reyes, M. (2017). Seeds of non-native species in King George Island soil. *Antarctic Science, 29*(4), Article 4. https://doi.org/10.1017/S0954102017000037

Fulgione, D., Rippa, D., Buglione, M., Trapanese, M., Petrelli, S., & Maselli, V. (2016). Unexpected but welcome. Artificially selected traits may increase fitness in wild boar. *Evolutionary Applications, 9*(6), 769–776. https://doi.org/10.1111/eva.12383

Furumo, R., & Aide, M. (2017). Characterizing commercial oil palm expansion in Latin America: Land use change and trade. *Environmental Research Letters, 12*(2), 024008. https://doi.org/10.1088/1748-9326/aa5892

Gaddum, M. (1999). *Gaddum's plant finder 2000 Where to buy 30,000 plants & seeds.* New Zealand Plant Finder.

Gaertner, M., Biggs, R., Te Beest, M., Hui, C., Molofsky, J., & Richardson, D. M. (2014). Invasive plants as drivers of regime shifts: Identifying high-priority invaders that alter feedback relationships. *Diversity and Distributions, 20*(7), 733–744. https://doi.org/10.1111/ddi.12182

Galil, B. S. (2006). The Marine Caravan – The Suez Canal and the Erythrean Invasion. In S. Gollasch, B. S. Galil, & A. N. Cohen (Eds.), *Bridging divides: Maritime canals as invasion corridors* (pp. 207–300). Springer Netherlands. https://doi.org/10.1007/978-1-4020-5047-3_6

Galil, B. S., Boero, F., Campbell, M. L., Carlton, J. T., Cook, E., Fraschetti, S., Gollasch, S., Hewitt, C. L., Jelmert, A., Macpherson, E., Marchini, A., McKenzie, C., Minchin, D., Occhipinti-Ambrogi, A., Ojaveer, H., Olenin, S., Piraino, S., & Ruiz, G. M. (2015). 'Double trouble': The expansion of the Suez Canal and marine bioinvasions in the Mediterranean Sea. *Biological Invasions, 17*(4), 973–976. https://doi.org/10.1007/s10530-014-0778-y

Gallardo, B. (2014). Europe's top 10 invasive species: Relative importance of climatic, habitat and socio-economic factors. *Ethology Ecology & Evolution, 26*(2–3), 130–151. https://doi.org/10.1080/03949370.2014.896417

Gallardo, B., & Aldridge, D. C. (2013). The 'dirty dozen': Socio-economic factors amplify the invasion potential of 12 high-risk aquatic invasive species in Great Britain and Ireland. *Journal of Applied Ecology, 50*(3), 757–766. https://doi.org/10.1111/1365-2664.12079

García, R. A., Franzese, J., Policelli, N., Sasal, Y., Zenni, R. D., Nuñez, M. A., Taylor, K., & Pauchard, A. (2018). Non-native Pines Are Homogenizing the Ecosystems of South America. In R. Rozzi, R. H. May, F. S. Chapin III, F. Massardo, M. C. Gavin, I. J. Klaver, A. Pauchard, M. A. Nuñez, & D. Simberloff (Eds.), *From Biocultural Homogenization to Biocultural Conservation* (Vol. 3, pp. 245–263). Springer International Publishing. https://doi.org/10.1007/978-3-319-99513-7_15

García-Díaz, P., Ross, J. V., Woolnough, A. P., & Cassey, P. (2016). The Illegal Wildlife Trade Is a Likely Source of Alien Species. *Conservation Letters, 10*(6), 690–698. https://doi.org/10.1111/conl.12301

Gavier-Pizarro, G. I., Radeloff, V. C., Stewart, S. I., Huebner, C. D., & Keuler, N. S. (2010). Housing is positively associated with invasive exotic plant species richness in New England, USA. *Ecological Applications, 20*(7), 1913–1925. https://doi.org/10.1890/09-2168.1

Gavira-O'Neill, K., Guerra-García, J. M., Moreira, J., & Ros, M. (2018). Mobile epifauna of the invasive bryozoan *Tricellaria inopinata*: Is there a potential invasional meltdown? *Marine Biodiversity, 48*(2), 1169–1178. https://doi.org/10.1007/s12526-016-0563-5

Ge, Z.-M., Cao, H.-B., Zhao, B., Zhang, L.-Q., & Cui, L.-F. (2015). Future vegetation patterns and primary production in the coastal wetlands of East China under sea level rise, sediment reduction, and saltwater intrusion. *Journal of Geophysical Research: Biogeoscience, 120*(10), 1923–1940. https://doi.org/10.1002/2015JG003014

Geheb, K., Kalloch, S., Medard, M., Nyapendi, A.-T., Lwenya, C., & Kyangwa, M. (2008). Nile perch and the hungry of Lake Victoria: Gender, status and food in an East African fishery. *Food Policy, 33*(1), 85–98. https://doi.org/10.1016/j.foodpol.2007.06.001

Geist, V. (1985). Game Ranching: Threat to Wildlife Conservation in North America. *Wildlife Society Bulletin (1973-2006), 13*(4), 594–598. https://www.jstor.org/stable/3782696

Gelbard, J. L., & Belnap, J. (2003). Roads as Conduits for Exotic Plant Invasions in a Semiarid Landscape. *Conservation Biology, 17*(2), 420–432. https://doi.org/10.1046/j.1523-1739.2003.01408.x

Gennaro, P., & Piazzi, L. (2011). Synergism between two anthropic impacts: *Caulerpa racemosa* var. *cylindracea* invasion and seawater nutrient enrichment. *Marine Ecology Progress Series, 427*, 59–70. https://doi.org/10.3354/meps09053

Genovesi, P., Bacher, S., Kobelt, M., Pascal, M., & Scalera, R. (2009). Alien Mammals of Europe. In *Handbook of Alien Species in Europe* (Vol. 3, pp. 119–128). Springer Netherlands. https://doi.org/10.1007/978-1-4020-8280-1_9

Geraldi, N. R., Smyth, A. R., Piehler, M. F., & Peterson, C. H. (2014). Artificial substrates enhance non-native macroalga and N_2 production. *Biological Invasions, 16*(9), 1819–1831. https://doi.org/10.1007/s10530-013-0629-2

Gerlach, J., Barker, G. M., Bick, C. S., Bouchet, P., Brodie, G., Christensen, C. C., Collins, T., Coote, T., Cowie, R. H., & Fiedler, G. C. (2021). Negative impacts of invasive predators used as biological control agents against the pest snail *Lissachatina fulica*: The snail *Euglandina 'rosea'* and the flatworm *Platydemus manokwari. Biological Invasions, 23*(4), 997–1031. https://doi.org/10.1007/s10530-020-02436-w

Germino, M. J., Belnap, J., Stark, J. M., Allen, E. B., & Rau, B. M. (2016). Ecosystem Impacts of Exotic Annual Invaders in the Genus *Bromus*. In M. J. Germino, J. C. Chambers, & C. S. Brown (Eds.), *Exotic Brome-Grasses in Arid and Semiarid Ecosystems of the Western US* (pp. 61–95). Springer International Publishing. https://doi.org/10.1007/978-3-319-24930-8_3

Gherardi, F., Gollasch, S., Minchin, D., Olenin, S., & Panov, V. (2009). Alien Invertebrates and Fish in European Inland Waters. In *Handbook of Alien Species in Europe* (Vol. 3, pp. 81–92). Springer. https://doi.org/10.1007/978-1-4020-8280-1_6

Gibbs, J., & Sheffield, C. S. (2009). Rapid Range Expansion of the Wool-Carder Bee, *Anthidium manicatum* (Linnaeus) (Hymenoptera: Megachilidae), in North America. *Journal of the Kansas Entomological Society, 82*(1), 21–29. https://doi.org/10.2317/JKES805.27.1

Gibson, P. P., Olden, J. D., & O'Neill, M. W. (2015). Beaver dams shift desert fish assemblages toward dominance by non-native species (Verde River, Arizona, USA). *Ecology of Freshwater Fish, 24*(3), 355–372. https://doi.org/10.1111/eff.12150

Gilliam, F. S. (2006). Response of the herbaceous layer of forest ecosystems to excess nitrogen deposition. *Journal of Ecology, 94*(6), Article 6. https://doi.org/10.1111/j.1365-2745.2006.01155.x

GISD. (2021). *Global Invasive Species Database*. http://www.iucngisd.org/gisd/

Glaesser, D., Kester, J., Paulose, H., Alizadeh, A., & Valentin, B. (2017). Global travel patterns: An overview. *Journal of Travel Medicine, 24*(4), tax007. https://doi.org/10.1093/jtm/tax007

Glasby, T. M., Connell, S. D., Holloway, M. G., & Hewitt, C. L. (2007). Nonindigenous biota on artificial structures: Could habitat creation facilitate biological invasions? *Marine Biology, 151*(3), 887–895. https://doi.org/10.1007/s00227-006-0552-5

Goedbloed, D. J., Megens, H. J., Hooft, P. V., Herrero-Medrano, J. M., Lutz, W., Alexandri, P., Crooijmans, R. P. M. A., Groenen, M., Wieren, S. E. V., Ydenberg, R. C., & Prins, H. H. T. (2013). Genome-wide single nucleotide polymorphism analysis reveals recent genetic introgression from domestic pigs into Northwest European wild boar populations. *Molecular Ecology, 22*(3), 856–866. https://doi.org/10.1111/j.1365-294X.2012.05670.x

Going, B. M., Hillerislambers, J., & Levine, J. M. (2009). Abiotic and biotic resistance to grass invasion in serpentine annual plant communities. *Oecologia, 159*(4), 839–847. https://doi.org/10.1007/s00442-008-1264-y

Golani, D. (2021). An updated Checklist of the Mediterranean fishes of Israel, with illustrations of recently recorded species and delineation of Lessepsian migrants. *Zootaxa, 4956*(1), Article 1. https://doi.org/10.11646/zootaxa.4956.1.1

Gong, X., Chen, Y., Wang, T., Jiang, X., Hu, X., & Feng, J. (2020). Double-edged effects of climate change on plant invasions: Ecological niche modeling global distributions of two invasive alien plants. *Science of The Total Environment, 740*, 139933. https://doi.org/10.1016/j.scitotenv.2020.139933

Goulson, D. (2003). Effects of Introduced Bees on Native Ecosystems. *Annual Review of Ecology, Evolution, and Systematics, 34*(1), 1–26. https://doi.org/10.1146/annurev.ecolsys.34.011802.132355

Government of the Republic of South Africa. (2016). *National Environmental Management: Biodiversity Act (10/2004): Alien and Invasive Species Lists*. https://www.gov.za/sites/default/files/gcis_document/201607/40166gon864.pdf

Gower, S. T. (2008). Are horses responsible for introducing non-native plants along forest trails in the eastern United States? *Forest Ecology and Management, 256*(5), 997–1003. https://doi.org/10.1016/j.foreco.2008.06.012

Gozlan, R. E. (2008). Introduction of non-native freshwater fish: Is it all bad? *Fish and Fisheries, 9*(1), 106–115. https://doi.org/10.1111/j.1467-2979.2007.00267.x

Grace, J. B., Smith, M. D., Grace, S. L., Collins, S. L., & Stohlgren, T. J. (2001). *Interactions between fire and invasive plants in temperate grasslands of North America*. 40–65. https://pubs.er.usgs.gov/publication/70208006

Gracia C., A., Rangel-Buitrago, N., & Florez, P. (2018). Beach litter and woody-debris colonizers on the Atlantico department Caribbean coastline, Colombia. *Marine Pollution Bulletin, 128*, 185–196. https://doi.org/10.1016/j.marpolbul.2018.01.017

Graham, C. A., Maron, M., & McAlpine, C. A. (2012). Influence of landscape structure on invasive predators: Feral cats and red foxes in the brigalow landscapes, Queensland, Australia. *Wildlife Research, 39*(8), 661–676. https://doi.org/10.1071/WR12008

Gravuer, K., Sullivan, J. J., Williams, P. A., & Duncan, R. P. (2008). Strong human association with plant invasion success for *Trifolium* introductions to New Zealand. *Proceedings of the National Academy of Sciences of the United States of America, 105*(17), 6344–6349. https://doi.org/10.1073/pnas.0712026105

Great Lakes Phragmites Collaborative. (2022). *Linking people, information & action.* https://www.greatlakesphragmites.net/

Green, P. T., O'Dowd, D. J., Abbott, K. L., Jeffery, M., Retallick, K., & Mac Nally, R. (2011). Invasional meltdown: Invader–invader mutualism facilitates a secondary invasion. *Ecology, 92*(9), 1758–1768. https://doi.org/10.1890/11-0050.1

Greet, J., Cousens, R. D., & Webb, J. A. (2013). More Exotic and Fewer Native Plant Species: Riverine Vegetation Patterns Associated with Altered Seasonal Flow Patterns. *River Research and Applications, 29*(6), 686–706. https://doi.org/10.1002/rra.2571

Gregoric, D. E. G., Beltramino, A. A., & Vogler, R. E. (2013). Expansión del rango de distribución de *Achatina fulica* Bowdich, 1822 (Gastropoda) en la Argentina y su concordancia con modelos predictivos. *Amici Molluscarum, 21*(1), 17–21. https://www.semanticscholar.org/paper/Expansi%C3%B3n-del-rango-de-distribuci%C3%B3n-de-Achatina-en-Gregoric-Beltramino/1adbc2d6ee8d8d9f519b025c57a3e5c4ea74a955

Greve, M., Mathakutha, R., Steyn, C., & Chown, S. L. (2017). Terrestrial invasions on sub-Antarctic Marion and Prince Edward Islands. *Bothalia, 47*(2). https://doi.org/10.4102/abc.v47i2.2143

Grez, A. A., Rand, T. A., Zaviezo, T., & Castillo-Serey, F. (2013). Land use intensification differentially benefits alien over native predators in agricultural landscape mosaics. *Diversity and Distributions, 19*(7), Article 7. https://doi.org/10.1111/ddi.12027

Griffin, A. S., Netto, K., & Peneaux, C. (2017). Neophilia, innovation and learning in an urbanized world: A critical evaluation of mixed findings. *Current Opinion in Behavioral Sciences, 16*, 15–22. https://doi.org/10.1016/j.cobeha.2017.01.004

Griffiths, C. J., Jones, C. J., Hansen, D. M., Puttoo, M., Tatayah, R. V., Müller, C. B., & Harris, S. (2010). The use of extant non-indigenous tortoises as a restoration tool to replace extinct ecosystem engineers. *Restoration Ecology, 18*(1), 1–7. https://doi.org/10.1111/j.1526-100X.2009.00612.x

Gritti, E. S., Smith, B., & Sykes, M. T. (2006). Vulnerability of Mediterranean Basin ecosystems to climate change and invasion by exotic plant species. *Journal of Biogeography, 33*(1), 145–157. https://doi.org/10.1111/j.1365-2699.2005.01377.x

Grosholz, E. D. (2002). Ecological and evolutionary consequences of coastal invasions. *Trends in Ecology & Evolution, 17*(1), 22–27. https://doi.org/10.1016/S0169-5347(01)02358-8

Grosholz, E. D. (2005). Recent biological invasion may hasten invasional meltdown by accelerating historical introductions. *Proceedings of the National Academy of Sciences, 102*(4), 1088–1091. https://doi.org/10.1073/pnas.0308547102

Grosholz, E. D., Crafton, R. E., Fontana, R. E., Pasari, J. R., Williams, S. L., & Zabin, C. J. (2015). Aquaculture as a vector for marine invasions in California. *Biological Invasions, 17*(5), 1471–1484. https://doi.org/10.1007/s10530-014-0808-9

Guay, P.-J., & Tracey, J. P. (2009). Feral Mallards: A risk for hybridisation with wild Pacific Black Ducks in Australia? *Victorian Naturalist, The, 126*(3), 87–91. https://search.informit.org/doi/epdf/10.3316/informit.669302995398699

Guerra-García, A., Golubov, J., & C. Mandujano, M. (2015). Invasion of *Kalanchoe* by clonal spread. *Biological Invasions, 17*(6), 1615–1622. https://doi.org/10.1007/s10530-014-0820-0

Guerrero, E. L. (2017). *Pteris nipponica* (Pteridaceae), un nueveo registro para la flora de buenos aires, Argentina. *Bonplandia, 26*(2), 137–142. JSTOR. https://doi.org/10.30972/bon.2622573

Guiden, P., Gorchov, D. L., Nielsen, C., & Schauber, E. (2015). Seed dispersal of an invasive shrub, Amur honeysuckle *(Lonicera maackii)*, by white-tailed deer in a fragmented agricultural-forest matrix. *Plant Ecology, 216*(7), 939–950. https://doi.org/10.1007/s11258-015-0480-x

Guo, Q., Fei, S., Shen, Z., Iannone, B. V., Knott, J., & Chown, S. L. (2018). A global analysis of elevational distribution of non-native *versus* native plants. *Journal of Biogeography, 45*(4), 793–803. https://doi.org/10.1111/jbi.13145

Guo, Q., Riitters, K., & Potter, K. (2018). A Subcontinental Analysis of Forest Fragmentation Effects on Insect and Disease Invasion. *Forests, 9*(12), 744. https://doi.org/10.3390/f9120744

Haeuser, E., Dawson, W., Thuiller, W., Dullinger, S., Block, S., Bossdorf, O., Carboni, M., Conti, L., Dullinger, I., Essl, F., Klonner, G., Moser, D., Münkemüller, T., Parepa, M., Talluto, M. V., Kreft, H., Pergl, J., Pyšek, P., Weigelt, P., … van Kleunen, M. (2018). European ornamental garden flora as an invasion debt under climate change. *Journal of Applied Ecology, 55*(5), Article 5. https://doi.org/10.1111/1365-2664.13197

Hager, H. A., Sinasac, S. E., Gedalof, Z., & Newman, J. A. (2014). Predicting Potential Global Distributions of Two *Miscanthus* Grasses: Implications for Horticulture, Biofuel Production, and Biological Invasions. *PLoS ONE, 9*(6), e100032. https://doi.org/10.1371/journal.pone.0100032

Haider, S., Kueffer, C., Bruelheide, H., Seipel, T., Alexander, J. M., Rew, L. J., Arévalo, J. R., Cavieres, L. A., McDougall, K. L., Milbau, A., Naylor, B. J., Speziale, K., & Pauchard, A. (2018). Mountain roads and non-native species modify elevational patterns of plant diversity. *Global Ecology and Biogeography, 27*(6), 667–678. https://doi.org/10.1111/geb.12727

Hajdu, S., Pertola, S., & Kuosa, H. (2005). *Prorocentrum minimum* (Dinophyceae) in the Baltic Sea: Morphology, occurrence—A review. *Harmful Algae, 4*(3), Article 3. https://doi.org/10.1016/j.hal.2004.08.004

Hales, J. M., & Fletcher, R. L. (1989). Studies on the recently introduced brown alga *Sargassum muticum* (Yendo) Fensholt. *Botanica Marina, 33*, 241–249. https://doi.org/10.1515/botm.1990.33.3.241

Hall-Spencer, J. M., Kelly, J., & Maggs, C. A. (2010). *Background document on maerl beds* (Vol. 491). OSPAR Commission, London. https://www.researchgate.net/publication/230661931_Background_document_on_maerl_beds

Hall-Spencer, J. M., Rodolfo-Metalpa, R., Martin, S., Ransome, E., Fine, M., Turner, S. M., Rowley, S. J., Tedesco, D., & Buia, M.-C. (2008). Volcanic carbon dioxide vents show ecosystem effects of ocean acidification. *Nature, 454*(7200), 96–99. https://doi.org/10.1038/nature07051

Hansen, D. M. (2015). Non-native megaherbivores: The case for novel function to manage plant invasions on islands. *AoB Plants, 7*, plv085. https://doi.org/doi: 10.1093/aobpla/plv085

Hansen, M. J., & Clevenger, A. P. (2005). The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation, 125(2), 249–259. https://doi.org/10.1016/j.biocon.2005.03.024

Hanski, I. (1999). Habitat connectivity, habitat continuity, and metapopulations in dynamic landscapes. *Oikos, 87*(2), 209–219. https://doi.org/10.2307/3546736

Hardwick, N. V., Armstrong, A. C., Ellis, S. A., Gladders, P., Yarham, D. J., & Parker, W. E. (1996). The impact of climate change on crop pests and diseases. *Aspects of Applied Biology, 45*, 261–268.

Haregeweyn, N., Tsunekawa, A., Tsubo, M., Meshesha, D., & Melkie, A. (2013). Analysis of the invasion rate, impacts and control measures of *Prosopis juliflora*: A case study of Amibara District, Eastern Ethiopia. *Environmental Monitoring and Assessment, 185*(9), 7527–7542. https://doi.org/10.1007/s10661-013-3117-3

Hassall, C. (2014). The ecology and biodiversity of urban ponds: Ecology and biodiversity of urban ponds. *Wiley Interdisciplinary Reviews: Water, 1*(2), 187–206. https://doi.org/10.1002/wat2.1014

Hattenschwiler, S., & Körner, C. (2003). Does elevated CO_2 facilitate naturalization of the non-indigenous *Prunus laurocerasus* in Swiss temperate forests? *Functional Ecology, 17*(6), 778–785. https://doi.org/10.1111/j.1365-2435.2003.00785.x

Havel, J. E., Kovalenko, K. E., Thomaz, S. M., Amalfitano, S., & Kats, L. B. (2015). Aquatic invasive species: Challenges for the future. *Hydrobiologia, 750*(1), Article 1. https://doi.org/10.1007/s10750-014-2166-0

Havel, J. E., Lee, C. E., & Vander Zanden, M. J. (2005). Do Reservoirs Facilitate Invasions into Landscapes? *BioScience, 55*(6), 518. https://doi.org/10.1641/0006-3568(2005)055[0518:DRFIIL]2.0.CO;2

Hays, W. S. T., & Conant, Sheila. (2007). Biology and Impacts of Pacific Island Invasive Species. 1. A Worldwide Review of Effects of the Small Indian Mongoose, *Herpestes javanicus* (Carnivora: Herpestidae). *Pacific Science, 61*(1), 3–16. https://doi.org/10.1353/psc.2007.0006

He, T., Lamont, B. B., & Pausas, J. G. (2019). Fire as a key driver of Earth's biodiversity. *Biological Reviews, 94*(6), Article 6. https://doi.org/10.1111/brv.12544

Hecky, R. E., Mugidde, R., Ramlal, P. S., Talbot, M. R., & Kling, G. W. (2010). Multiple stressors cause rapid ecosystem change in Lake Victoria. *Freshwater Biology, 55*(s1), 19–42. https://doi.org/10.1111/j.1365-2427.2009.02374.x

Heimpel, G. E., Frelich, L. E., Landis, D. A., Hopper, K. R., Hoelmer, K. A., Sezen, Z., Asplen, M. K., & Wu, K. (2010). European buckthorn and Asian soybean aphid as components of an extensive invasional meltdown in North America. *Biological Invasions, 12*(9), 2913–2931. https://doi.org/10.1007/s10530-010-9736-5

Hellmann, J. J., Byers, J. E., Bierwagen, B. G., & Dukes, J. S. (2008). Five Potential Consequences of Climate Change for Invasive Species. *Conservation Biology, 22*(3), 534–543. https://doi.org/10.1111/j.1523-1739.2008.00951.x

Henderson, L. (2009, July). Features used in identification of Eucalyptus species. *Sapia News (Southern African Plant Invaders Atlas), 12*, 3. https://www.midlandsconservancies.org.za/documents/problemplants/sapia/SAPIA_NEWS_No._12.pdf

Herbert, R. J. H., Humphreys, J., Davies, Clare. J., Roberts, C., Fletcher, S., & Crowe, Tasman. P. (2016). Ecological impacts of non-native Pacific oysters *(Crassostrea gigas)* and management measures for protected areas in Europe. *Biodiversity and Conservation, 25*(14), 2835–2865. https://doi.org/10.1007/s10531-016-1209-4

Herrando-Moraira, S., Vitales, D., Nualart, N., Gómez-Bellver, C., Ibáñez, N., Massó, S., Cachón-Ferrero, P., González-Gutiérrez, P. A., Guillot, D., Herrera, I., Shaw, D., Stinca, A., Wang, Z., & López-Pujol, J. (2020). Global distribution patterns and niche modelling of the invasive *Kalanchoe × houghtonii* (Crassulaceae). *Scientific Reports, 10*(1), Article 1. https://doi.org/10.1038/s41598-020-60079-2

Hickey, V. (2010). The quagga mussel crisis at Lake Mead National Recreation Area, Nevada (U.S.A.). Conservation Biology: *The Journal of the Society for Conservation Biology, 24*(4), 931–937. https://doi.org/10.1111/j.1523-1739.2010.01490.x

Hilton, M., Harvey, N., Hart, A., James, K., & Arbuckle, C. (2006). The Impact of Exotic Dune Grass Species on Foredune Development in Australia and New Zealand: A case study of *Ammophila arenaria* and *Thinopyrum junceiforme*. *Australian Geographer, 37*(3), 313–334. https://doi.org/10.1080/00049180600954765

Hindar, K., Hole, L. R., Kausrud, K. L., Malmstrøm, M., Rimstad, E., Robertson, L., Sandlund, O. T., Thorstad, E. B., Vollset, K., & de Boer, H. (2020). *Assessment of the risk to Norwegian biodiversity and aquaculture from pink salmon (Oncorhynchus gorbuscha). Scientific Opinion of the Panel on Alien Organisms and Trade in Endangered Species of the Norwegian Scientific Committee for Food and Environment (VKM report 2020:0). Norwegian Scientific Committee for Food and Environment* (VKM). https://vetinst.brage.unit.no/vetinst-xmlui/bitstream/handle/11250/2828612/HindarAssessmentVKMrapport01_2020.pdf?sequence=2

Hinz, H. L., Winston, R. L., & Schwarzländer, M. (2019). How Safe Is Weed Biological Control? A Global Review of Direct Nontarget Attack. *The Quarterly Review of Biology, 94*(1), 27. https://doi.org/10.1086/702340

Hirsch, H., Allsopp, M. H., Canavan, S., Cheek, M., Geerts, S., Geldenhuys, C. J., Harding, G., Hurley, B. P., Jones, W., Keet, J.-H., Klein, H., Ruwanza, S., van Wilgen, B. W., Wingfield, M. J., & Richardson, D. M. (2020). *Eucalyptus camaldulensis* in South Africa – past, present, future. *Transactions of the Royal Society of South Africa, 75*(1), 1–22. https://doi.org/10.1080/0035919X.2019.1669732

Hixon, M., Green, S., Albins, M., Akins, J., & Morris, J. (2016). Lionfish: A major marine invasion. *Marine Ecology Progress Series, 558*, 161–165. https://doi.org/10.3354/meps11909

Hobbs, R. J., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V. A., Epstein, P. R., Ewel, J. J., Klink, C. A., Lugo, A. E., Norton, D., Ojima, D., Richardson, D. M., Sanderson, E. W., Valladares, F., Vilà, M., Zamora, R., & Zobel, M. (2006). Novel ecosystems: Theoretical and management aspects of the new ecological world order: Novel ecosystems. *Global Ecology and Biogeography, 15*(1), 1–7. https://doi.org/10.1111/j.1466-822X.2006.00212.x

Hobbs, R. J., Higgs, E., & Harris, J. A. (2009). Novel ecosystems: Implications for conservation and restoration. *Trends in Ecology & Evolution, 24*(11), 599–605. https://www.sciencedirect.com/science/article/abs/pii/S0169534709002018

Hoegh-Guldberg, O., & Bruno, J. F. (2010). The Impact of Climate Change on the World's Marine Ecosystems. *Science, 328*(5985), 1523–1528. https://doi.org/10.1126/science.1189930

Hoegh-Guldberg, O., Hughes, L., McIntyre, S., Lindenmayer, D. B., Parmesan, C., Possingham, H. P., & Thomas, C. D. (2008). Assisted colonization and rapid climate change. *SCIENCE, 321*(5887), 345–346. https://doi.org/10.1126/science.1157897

Hoegh-Guldberg, O., Jacob, D., Taylor, M., Bindi, M., Brown, S., Camilloni, I., Diedhiou, A., Djalante, R., Ebi, K., Engelbrecht, F., Guiot, J., Hijioka, Y., Mehrotra, S., Payne, A., Seneviratne, S. I., Thomas, A., Warren, R., Zhou, G., Halim, S. A., … Zougmoré, R. (2018). Impacts of 1.5 oC global warming on natural and human systems. In *Global Warming of 1.5 °C. An IPCC special report on the impacts of global warming of 1.5 °C above preindustrial levels and related global greenhouse gas emission pathways.* https://helda.helsinki.fi/handle/10138/311749

Hoenicka, H., & Fladung, M. (2006). Biosafety in *Populus* spp. And other forest trees: From non-native species to taxa derived from traditional breeding and genetic engineering. *Trees, 20*(2), 131–144. https://doi.org/10.1007/s00468-005-0023-5

Holbech, H., & Pedersen, K. L. (2018). Ballast water and invasive species in the Arctic. In *Arctic marine resource governance and development* (pp. 115–137). http://www.vliz.be/en/search-persons?module=ref&refid=292858

Hong, J., Lee, G.-S., Park, J.-J., Mo, H., & Cho, K. (2019). Risk map for the range expansion of *Thrips palmi* in Korea under climate change: Combining species distribution models with land-use change. *Journal of Asia-Pacific Entomology, 22*(3), 666–674. https://doi.org/10.1016/j.aspen.2019.04.013

Hopkinson, R. F. (1999). Weather Patterns and the Occurrence of Diamondback Moths on the Canadian Prairies 1998. Environment Canada, Atmospheric & Hydrological Sciences Division. *Atmospheric Environment Branch Report, 71p.*

Hoppenreijs, J. H. T., Beringen, R., Collas, F. P. L., Eeuwes, D. D. M., Odé, B., & Leuven, R. (2019). *Risicobeoordeling van voedselbosbouw als introductieroute voor invasieve plantensoorten* (Rapport Afdeling Dierecologie En Fysiologie ; 2019-2, p. 193p) [External research report]. Nijmegen : Radboud Universiteit, Floron & NVWA. https://hdl.handle.net/2066/215963

Horton, T. R. (2017). Spore Dispersal in Ectomycorrhizal Fungi at Fine and Regional Scales. In L. Tedersoo (Ed.), *Biogeography of Mycorrhizal Symbiosis* (pp. 61–78). Springer International Publishing. https://doi.org/10.1007/978-3-319-56363-3_3

Hou, X., Liu, S., Cheng, F., Zhang, Y., Dong, S., Su, X., & Liu, G. (2019). Vegetation community composition along disturbance gradients of four typical open-pit mines in Yunnan Province of southwest China. *Land Degradation & Development, 30*(4), 437–447. https://doi.org/10.1002/ldr.3234

Howard, W. E., & Amaya, J. N. (1975). European rabbit invades western Argentina. *The Journal of Wildlife Management, 39*(4), 757–761. https://doi.org/10.2307/3800238

Hradsky, B. A., Mildwaters, C., Ritchie, E. G., Christie, F., & Di Stefano, J. (2017). Responses of invasive predators and native prey to a prescribed forest fire. *Journal of Mammalogy, 98*(3), 835–847. https://doi.org/10.1093/jmammal/gyx010

Hughes, K. A., Pescott, O. L., Peyton, J. M., Adriaens, T., Cottier-Cook, E. J., Key, G., Rabitsch, W., Tricarico, E., Barnes, D. K. A., Baxter, N., Belchier, M., Blake, D., Convey, P., Dawson, W., Frohlich, D., Gardiner, L. M., González-Moreno, P., James, R., Malumphy, C., … Roy, H. E. (2020). Invasive non-native species likely to threaten biodiversity and ecosystems in the Antarctic Peninsula region. *Global Change Biology, 26*(4), 2702–2716. https://doi.org/10.1111/gcb.14938

Huiskes, A. H. L., Gremmen, N. J. M., Bergstrom, D. M., Frenot, Y., Hughes, K. A., Imura, S., Kiefer, K., Lebouvier, M., Lee, J. E., Tsujimoto, M., Ware, C., Van de Vijver, B., & Chown, S. L. (2014). Aliens in Antarctica: Assessing transfer of plant propagules by human visitors to reduce invasion risk. *Biological Conservation, 171*, 278–284. https://doi.org/10.1016/j.biocon.2014.01.038

Hulme, P. E. (2006). Beyond control: Wider implications for the management of biological invasions. *Journal of Applied Ecology, 43*(5), 835–847. https://doi.org/10.1111/j.1365-2664.2006.01227.x

Hulme, P. E. (2009). Trade, transport and trouble: Managing invasive species pathways in an era of globalization. *Journal of Applied Ecology, 46*(1), 10–18. https://doi.org/10.1111/j.1365-2664.2008.01600.x

Hulme, P. E. (2011a). Addressing the threat to biodiversity from botanic gardens. *Trends in Ecology and Evolution, 26*(4), 168–174. https://doi.org/10.1016/j.tree.2011.01.005

Hulme, P. E. (2011b). Contrasting impacts of climate-driven flowering phenology on changes in alien and native plant species distributions. *New Phytologist, 189*(1), 272–281. https://doi.org/10.1111/j.1469-8137.2010.03446.x

Hulme, P. E. (2015a). Invasion pathways at a crossroad: Policy and research challenges for managing alien species introductions. *Journal of Applied Ecology, 52*(6), 1418–1424. https://doi.org/10.1111/1365-2664.12470

Hulme, P. E. (2015b). Resolving whether botanic gardens are on the road to conservation or a pathway for plant invasions. *Conservation Biology, 29*(3), 816–824. https://doi.org/10.1111/cobi.12426

Hulme, P. E. (2017). Climate change and biological invasions: Evidence, expectations, and response options. *Biological Reviews, 92*(3), 1297–1313. https://doi.org/10.1111/brv.12282

Hulme, P. E. (2020). Plant invasions in New Zealand: Global lessons in prevention, eradication and control. *Biological Invasions, 22*(5), 1539–1562. https://doi.org/10.1007/s10530-020-02224-6

Hulme, P. E. (2021a). Advancing One Biosecurity to Address the Pandemic Risks of Biological Invasions. *BioScience, 71*(7), 708–721. https://doi.org/10.1093/biosci/biab019

Hulme, P. E. (2021b). Unwelcome exchange: International trade as a direct and indirect driver of biological invasions worldwide. *One Earth, 4*(5), 666–679. https://doi.org/10.1016/j.oneear.2021.04.015

Hulme, P. E. (2022). Importance of greater interdisciplinarity and geographic scope when tackling the driving forces behind biological invasions. *Conservation Biology, 36*(2), e13817. https://doi.org/10.1111/cobi.13817

Hulme, P. E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., Nentwig, W., Olenin, S., Panov, V., Pergl, J., Pyšek, P., Roques, A., Sol, D., Solarz, W., & Vilà, M. (2008). Grasping at the routes of biological invasions: A framework for integrating pathways into policy. *Journal of Applied Ecology, 45*(2), 403–414. https://doi.org/10.1111/j.1365-2664.2007.01442.x

Hulme, P. E., Brundu, G., Carboni, M., Dehnen-Schmutz, K., Dullinger, S., Early, R., Essl, F., González-Moreno, P., Groom, Q. J., Kueffer, C., Kühn, I., Maurel, N., Novoa, A., Pergl, J., Pyšek, P., Seebens, H., Tanner, R., Touza, J. M., van Kleunen, M., & Verbrugge, L. N. H. (2018). Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions. *Journal of Applied Ecology, 55*(1), 92–98. https://doi.org/10.1111/1365-2664.12953

Hulme, P. E., Pyšek, P., & Winter, M. (2012). Biosecurity on Thin Ice in Antarctica. *Science, 336*(6085), 1101–1104. https://doi.org/10.1126/science.336.6085.1102-b

Humair, F., Humair, L., Kuhn, F., & Kueffer, C. (2015). E-commerce trade in invasive plants. *Conservation Biology, 29*(6), 1658–1665. https://doi.org/10.1111/cobi.12579

Hurrell, J., Cabanillas, P., Costantino, F. B., & Delucchi, G. (2012). Bignoniaceae adventicias en la Argentina. Primera cita de *Podranea ricasoliana* y nuevos registros de *Campsis radicans*. *Revista Del Museo Argentino de Ciencias Naturales Nueva Serie, 14*(1), Article 1. http://revista.macn.gov.ar/ojs/index.php/RevMus/article/view/207

Hutchings, S. (2003). The diet of feral house cats *(Felis catus)* at a regional rubbish tip, Victoria. *Wildlife Research, 30*(1), 103–110. https://doi.org/10.1071/wr99067

IAATO. (2020). *Issued by the International Association of Antarctica Tour Operators.* https://iaato.org/wp-content/uploads/2020/07/IAATO-on-Antarctic-visitor-figures-2019-20-FINAL.pdf

Iacarella, J. C., Davidson, I. C., & Dunham, A. (2019). Biotic exchange from movement of 'static' maritime structures. *Biological Invasions, 21*(4), 1131–1141. https://doi.org/10.1007/s10530-018-1888-8

Iannone, B. V., Potter, K. M., Hamil, K.-A. D., Huang, W., Zhang, H., Guo, Q., Oswalt, C. M., Woodall, C. W., & Fei, S. (2016). Evidence of biotic resistance to invasions in forests of the Eastern USA. *Landscape Ecology, 31*(1), 85–99. https://doi.org/10.1007/s10980-015-0280-7

Ibáñez, I., Silander Jr, J. A., Wilson, A. M., LaFleur, N., Tanaka, N., & Tsuyama, I. (2009). Multivariate forecasts of potential distributions of invasive plant species. *Ecological Applications, 19*(2), 359–375.

Imboma, T. S. (2014). *Ecological effects of the invasive House Sparrow (Passer domesticus) on sympatric native Sparrows in three urban areas in Kenya* [Masters Thesis]. University of Nairobi.

IMF. (2021). *Report for Selected Countries and Subjects: October 2021*. https://www.imf.org/en/Publications/WEO/weo-database/2021/October/weo-repor

International Energy Agency (IEA). (2020). *World energy outlook 2020*. OECD Publishing. https://www.iea.org/reports/world-energy-outlook-2020

International Groundwater Resources Assessment Centre. (2018). Groundwater Overview: Making the invisible Visible. *UN WATER.*

International Organization for Migration. (2019). *World migration report 2020.* IOM.

International Telecommunication Union. (2021). *Measuring digital development: Facts and figures 2021*. https://www.itu.int/en/ITU-D/Statistics/Pages/facts/default.aspx

IPBES. (2016a). *The assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3402857

IPBES. (2016b). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3235428

IPBES. (2018a). *Summary for policymakers of the regional assessment report on biodiversity and ecosystem services for Europe and Central Asia of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* (summary for policy makers, p. 48). IPBES secretariat. https://doi.org/10.5281/ZENODO.3237467

IPBES. (2018b). *The IPBES assessment report on land degradation and restoration.* (p. 744). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237392

IPBES. (2018c). *The IPBES regional assessment report on biodiversity and ecosystem services for Africa.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://zenodo.org/record/3236178#.YmjyldrP2Uk

IPBES. (2018d). *The IPBES regional assessment report on biodiversity and ecosystem services for Asia and the Pacific* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237374

IPBES. (2018e). *The IPBES regional assessment report on biodiversity and ecosystem services for Europe and Central Asia* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237429

IPBES. (2018f). *The IPBES regional assessment report on biodiversity and ecosystem services for the Americas.* Zenodo. https://doi.org/10.5281/zenodo.3236253

IPBES. (2019). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3831673

IPBES. (2020). *Report of the Indigenous and local knowledge dialogue workshop on the first order draft of the IPBES assessment of invasive alien species* (J. L. Andreve, R. Batzin Chojoj, A. Black, J. T. Cleofe, F. Daguitan, C. Grant, J. A. Guillao, L. Jacobs, T. Malcolm, L. Mulenkei, K. Kumar Rai, A. Nzovu, J. M. Ole Kaunga, M. E. Regpala, N. Sall, P. Shulbaeva, R. Spencer, P. Timoti, & Y. Upun, Eds.). https://ipbes.net/sites/default/files/inline-files/IPBES_IAS_2ndILKDialogue_FOD_Report_FINAL_ForWeb.pdf

IPBES. (2022a). *Methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.6522522

IPBES. (2022b). *Report of the third Indigenous and local knowledge dialogue workshop for the IPBES thematic assessment of invasive alien species and their control.* https://ipbes.net/sites/default/files/2023-02/IPBES_IAS_3rdILKDialogue_SPM-SOD_Report_FinalForWeb2.pdf

IPBES. (2022c). *Thematic assessment of the sustainable use of wild species of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.* Zenodo. https://doi.org/10.5281/ZENODO.6448568

IPCC. (2007). *Climate Change 2007: Synthesis report. Contribution of Working Groups 1, II, and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* (R. K. Pachauri & A. Reisinger, Eds.). IPCC, Geneva, Switzerland. https://www.ipcc.ch/

IPCC. (2014). Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. In R. K. Pachauri & M. R. Allen (Eds.), *EPIC3Geneva, Switzerland, IPCC, 151 p., pp. 151, ISBN: 978-92-9169-143-2* (p. 151pp). IPCC. https://epic.awi.de/id/eprint/37530/

IPCC. (2021). Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. *Cambridge University Press*. https://doi.org/10.1017/9781009157896

IPCC. (2022). *Climate Change 2022: Impacts, Adaptation and Vulnerability* (p. 3056) [Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change]. Cambridge University Press. 10.1017/9781009325844

IPCC. (2019). *Changing Ocean, Marine Ecosystems, and Dependent Communities—Special Report on the Ocean and Cryosphere in a Changing Climate.* https://www.ipcc.ch/srocc/chapter/chapter-5/

Iriarte, J. A., Lobos, G. A., & Jaksic, F. M. (2005). Invasive vertebrate species in Chile and their control and monitoring by governmental agencies. *Revista Chilena de Historia Natural, 78*(1), 143–154. https://www.scopus.com/inward/record.uri?eid=2-s2.0-19844372796&partnerID=40&md5=60b90e72810e001a8f4ebd8f0693954c

IRP. (2019). *Global Resources Outlook 2019: Natural Resources for the Future We Want.* Oberle, B., Bringezu, S., Hatfeld-Dodds, S., Hellweg, S., Schandl, H., Clement, J., and Cabernard, L., Che, N., Chen, D., Droz-Georget, H., Ekins, P., Fischer-Kowalski, M., Flörke, M., Frank, S., Froemelt, A., Geschke, A., Haupt, M., Havlik, P., Hüfner, R., Lenzen, M., Lieber, M., Liu, B., Lu, Y., Lutter, S., Mehr, J., Miatto, A., Newth, D., Oberschelp, C., Obersteiner, M., Pfster, S., Piccoli, E., Schaldach, R., Schüngel, J., Sonderegger, T., Sudheshwar, A., Tanikawa, H., van der Voet, E., Walker, C., West, J., Wang, Z., Zhu, B. A Report of the International Resource Panel. UNEP. https://orbi.uliege.be/handle/2268/244276

Isaak, D. J., Luce, C. H., Rieman, B. E., Nagel, D. E., Peterson, E. E., Horan, D. L., Parkes, S., & Chandler, G. L. (2010). Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. *Ecological Applications, 20*(5), 1350–1371. https://doi.org/10.1890/09-0822.1

ISAC. (2016). *Invasive species impacts on infrastructure* (p. 12). Invasive Species Advisory ommittee. https://www.doi.gov/sites/doi.gov/files/uploads/isac_infrastructure_white_paper.pdf

IUCN. (2020, September). *Motion 67—Reducing the impacts of the mining industry on biodiversity.* IUCN World Conservation Congress 2020. https://www.iucncongress2020.org/motion/067

Iverson, L. R., Rebbeck, J., Peters, M. P., Hutchinson, T., & Fox, T. (2019). Predicting *Ailanthus altissima* presence across a managed forest landscape in southeast Ohio. *Forest Ecosystems, 6*(1), 41. https://doi.org/10.1186/s40663-019-0198-7

Ivkić, A., Steger, J., Galil, B. S., & Albano, P. G. (2019). The potential of large rafting objects to spread Lessepsian invaders: The case of a detached buoy. *Biological Invasions, 21*(6), 1887–1893. https://doi.org/10.1007/s10530-019-01972-4

Izquierdo-Gómez, D., Arechavala-Lopez, P., Bayle-Sempere, J. T., & Sánchez-Jerez, P. (2017). Assessing the influence of gilthead sea bream escapees in landings of Mediterranean fisheries through a scale-based methodology. *Fisheries Management and Ecology, 24*(1), 62–72. https://doi.org/10.1111/fme.12202

Jackson, M. C., Loewen, C. J. G., Vinebrooke, R. D., & Chimimba, C. T. (2016). Net effects of multiple stressors in freshwater ecosystems: A meta-analysis. *Global Change Biology, 22*(1), 180–189. https://doi.org/10.1111/gcb.13028

Jactel, H., Desprez-Loustau, M.-L., Battisti, A., Brockerhoff, E., Santini, A., Stenlid, J., Björkman, C., Branco, M., Dehnen-Schmutz, K., Douma, J. C., Drakulic, J., Drizou, F., Eschen, R., Franco, J. C., Gossner, M. M., Green, S., Kenis, M., Klapwijk, M. J., Liebhold, A. M., … Zalucki, M. P. (2020). Pathologists and entomologists must join forces against forest pest and pathogen invasions. *NeoBiota, 58*, 107–127. https://doi.org/10.3897/neobiota.58.54389

Jauni, M., Gripenberg, S., & Ramula, S. (2015). Non-native plant species benefit from disturbance: A meta-analysis. *Oikos, 124*(2), 122–129. https://doi.org/10.1111/oik.01416

Jaureguiberry, P., Titeux, N., Wiemers, M., Bowler, D. E., Coscieme, L., Golden, A. S., Guerra, C. A., Jacob, U., Takahashi, Y., Settele, J., Díaz, S., Molnár, Z., & Purvis, A. (2022). The direct drivers of recent global anthropogenic biodiversity loss. *Science Advances, 8*(45), eabm9982. https://doi.org/10.1126/sciadv.abm9982

Jentsch, A., Kreyling, J., & Beierkuhnlein, C. (2007). A new generation of climate-change experiments: Events, not trends. *Frontiers in Ecology and the Environment, 5*(7), 365–374. https://doi.org/10.1890/1540-9295(2007)5[365:ANGOCE]2.0.CO;2

Jerde, C. L., Chadderton, W. L., Mahon, A. R., Renshaw, M. A., Corush, J., Budny, M. L., Mysorekar, S., & Lodge, D. M. (2013). Detection of Asian carp DNA as part of a Great Lakes basin-wide surveillance program. *Canadian Journal of Fisheries and Aquatic Sciences, 70*(4), 522–526. https://doi.org/10.1139/cjfas-2012-0478

Jeschke, J. M., Aparicio, L. G., Haider, S., Heger, T., Lortie, C., Pyšek, P., & Strayer,

D. (2012). Support for major hypotheses in invasion biology is uneven and declining. *NeoBiota, 14*, 1–20. https://doi.org/10.3897/neobiota.14.3435

Jeschke, J. M., & Genovesi, P. (2011). Do biodiversity and human impact influence the introduction or establishment of alien mammals? *Oikos, 120*(1), 57–64. https://doi.org/10.1111/j.1600-0706.2010.18621.x

Jeschke, J. M., & Strayer, D. L. (2005). Invasion success of vertebrates in Europe and North America. *Proceedings of the National Academy of Sciences, 102*(20), 7198–7202. https://doi.org/10.1073/pnas.0501271102

Jevon, T., & Shackleton, C. M. (2015). Integrating Local Knowledge and Forest Surveys to Assess *Lantana camara* Impacts on Indigenous Species Recruitment in Mazeppa Bay, South Africa. *Human Ecology, 43*(2), 247–254. https://doi.org/10.1007/s10745-015-9748-y

Jiménez, M. A., Jaksic, F. M., Armesto, J. J., Gaxiola, A., Meserve, P. L., Kelt, D. A., & Gutiérrez, J. R. (2011). Extreme climatic events change the dynamics and invasibility of semi-arid annual plant communities: Extreme events and plant invasibility. *Ecology Letters, 14*(12), 1227–1235. https://doi.org/10.1111/j.1461-0248.2011.01693.x

Jodoin, Y., Lavoie, C., Villeneuve, P., Theriault, M., Beaulieu, J., & Belzile, F. (2008). Highways as corridors and habitats for the invasive common reed *Phragmites australis* in Quebec, Canada. *Journal of Applied Ecology, 45*(2), 459–466. https://doi.org/10.1111/j.1365-2664.2007.01362.x

Johnson, J., Bell, J., & Gupta, A. S. (2016). *Pacific islands ocean acidification vulnerability assessment*. https://www.sprep.org/attachments/Publications/CC/ocean-acidification.pdf

Johnson, P. T., Olden, J. D., & Zanden, M. J. V. (2008). Dam invaders: Impoundments facilitate biological invasions into freshwaters. *Frontiers in Ecology and the Environment, 6*(7), 357–363. https://doi.org/10.1890/070156

Johnston, M., & Purkis, S. (2015). Hurricanes accelerated the Florida-Bahamas lionfish invasion. *Global Change Biology, 21*(6), 2249–2260. https://doi.org/10.1111/gcb.12874

Jones, A. E., Turner, J., Caminade, C., Heath, A. E., Wardeh, M., Kluiters, G., Diggle, P. J., Morse, A. P., & Baylis, M. (2019). Bluetongue risk under future climates. *Nature Climate Change, 9*(2), 153–157. https://doi.org/10.1038/s41558-018-0376-6

Jones, C., Barron, M., Warburton, B., Coleman, M., Lyver, P. O., & Nugent, G. (2012). Serving two masters: Reconciling economic and biodiversity outcomes of brushtail possum *(Trichosurus vulpecula)* fur harvest in an indigenous New Zealand forest. *Biological Conservation, 153*, 143–152. https://doi.org/10.1016/j.biocon.2012.04.016

Jørstad, K. E., Otterå, H., van der Meeren, T., Dahle, G., Paulsen, O. I., Bakke, G., & Svåsand, T. (2014). Genetic marking of farmed Atlantic cod (*Gadus morhua* l.) and detection of escapes from a commercial cod farm. *ICES Journal of Marine Science, 71*(3), 574–584. https://doi.org/10.1093/icesjms/fst171

Jouffray, J.-B., Blasiak, R., Norström, A. V., Österblom, H., & Nyström, M. (2020). The Blue Acceleration: The Trajectory of Human Expansion into the Ocean. *One Earth, 2*(1), 43–54. https://doi.org/10.1016/j.oneear.2019.12.016

Kaique, L. P. M., & Nara, L. B. C. (2017). Occurrence of *Achatina fulica* Bowdich, 1822 (Mollusca, Gas-tropoda) and its correlation with vegetation and garbage found in São Mateus, Espírito Santo, Brazil. *Natureza on Line, 15*(1), 28–32.

Kalwij, J. M., Robertson, M. P., & van Rensburg, B. J. (2015). Annual monitoring reveals rapid upward movement of exotic plants in a montane ecosystem. *Biological Invasions, 17*(12), 3517–3529. https://doi.org/10.1007/s10530-015-0975-3

Kapitza, K., Zimmermann, H., Martín-López, B., & Von Wehrden, H. (2019). Research on the social perception of invasive species: A systematic literature review. *NeoBiota, 43*, 47–68. https://doi.org/10.3897/neobiota.43.31619

Karatayev, A. Y., Mastitsky, S. E., Burlakova, L. E., & Olenin, S. (2008). Past, current, and future of the central European corridor for aquatic invasions in Belarus. *Biological Invasions, 10*(2), 215–232. https://doi.org/10.1007/s10530-007-9124-y

Karlsson, S., Diserud, O. H., Fiske, P., Hindar, K., & Handling editor: W. Stewart Grant. (2016). Widespread genetic introgression of escaped farmed Atlantic salmon in wild salmon populations. *ICES Journal of Marine Science, 73*(10), 2488–2498. https://doi.org/10.1093/icesjms/fsw121

Kastner, T., Erb, K.-H., & Nonhebel, S. (2011). International wood trade and forest change: A global analysis. *Global Environmental Change-Human and Policy Dimensions, 21*(3), 947–956. https://doi.org/10.1016/j.gloenvcha.2011.05.003

Katsanevakis, S., Zenetos, A., Belchior, C., & Cardoso, A. C. (2013). Invading European Seas: Assessing pathways of introduction of marine aliens. *Ocean & Coastal Management, 76*, 64–74. https://doi.org/10.1016/j.ocecoaman.2013.02.024

Kaza, S., Yao, L., Bhada-Tata, P., & Van Woerden, F. (2018). *What a waste 2.0: A global snapshot of solid waste management to 2050.* The World Bank.

Keeley, J. E., Lubin, D., & Fotheringham, C. J. (2003). Fire and grazing impacts on plant diversity and alien plant invasions in the southern Sierra Nevada. *Ecological Applications, 13*(5), 1355–1374. https://doi.org/10.1890/02-5002

Keeley, J. E., Pausas, J. G., Rundel, P. W., Bond, W. J., & Bradstock, R. A. (2011). Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science, 16*(8), Article 8. https://doi.org/10.1016/j.tplants.2011.04.002

Kelly, J. (2018). The introduction of the Nile Perch into Lake Victoria and the resulting neo-colonial relationship. *Undergraduate Journal of Global Citizenship, 2*(4), 14. https://digitalcommons.fairfield.edu/jogc/vol2/iss4/3

Kelly, L. T., Giljohann, K. M., Duane, A., Aquilué, N., Archibald, S., Batllori, E., Bennett, A. F., Buckland, S. T., Canelles, Q., Clarke, M. F., Fortin, M.-J., Hermoso, V., Herrando, S., Keane, R. E., Lake, F. K., McCarthy, M. A., Morán-Ordóñez, A., Parr, C. L., Pausas, J. G., … Brotons, L. (2020). Fire and biodiversity in the Anthropocene. *Science, 370*(6519), eabb0355. https://doi.org/10.1126/science.abb0355

Kempel, A., Chrobock, T., Fischer, M., Rohr, R. P., & Kleunen, M. van. (2013). Determinants of plant establishment success in a multispecies introduction experiment with native and alien species. *Proceedings of the National Academy of Sciences, 110*(31), 12727–12732. https://doi.org/10.1073/pnas.1300481110

Kerckhof, F., De Mesel, I., & Degraer, S. (2016). Do wind farms favour introduced hard substrata species? In S. Degraer, R. Brabant, B. Rumes, & L. Vigin (Eds.), *Environmental impacts of offshore wind farms in the Belgian part of the North Sea* (pp. 61–75). Royal Belgian Institute of Natural Sciences. https://core.ac.uk/download/pdf/80852935.pdf

Kerns, B. K., Tortorelli, C., Day, M. A., Nietupski, T., Barros, A. M. G., Kim, J. B., & Krawchuk, M. A. (2020). Invasive grasses: A new perfect storm for forested ecosystems? *Forest Ecology and Management, 463*, 117985. https://doi.org/10.1016/j.foreco.2020.117985

Khuroo, A. A., Rashid, I., Reshi, Z., Dar, G., & Wafai, B. (2007). The alien flora of Kashmir Himalaya. *Biological Invasions, 9*(3), 269–292.

Kiessling, T., Gutow, L., & Thiel, M. (2015). Marine Litter as Habitat and Dispersal Vector. In Bergmann, M and Gutow, L and Klages, M (Ed.), *Marine Anthropogenic Litter* (pp. 141–181). Springer-Verlag, Berlin. https://doi.org/10.1007/978-3-319-16510-3_6

Kilungu, H., Leemans, R., Munishi, P. K., Nicholls, S., & Amelung, B. (2019). Forty Years of Climate and Land-Cover Change and its Effects on Tourism Resources in Kilimanjaro National Park. *Tourism Planning & Development, 16*(2), 235–253. https://doi.org/10.1080/21568316.2019.1569121

Kimbro, D. L., Cheng, B. S., & Grosholz, E. D. (2013). Biotic resistance in marine environments. *Ecology Letters, 16*, 821–833. https://doi.org/10.1111/ele.12106

King, C. M., & Barrett, P. (2005). *The Handbook of New Zealand Mammals* (Second edition). Oxford University Press. https://natlib.govt.nz/records/22051977?search%5Bpath%5D=items&search%5Btext%5D=Jones%252C+Carolyn

King, C. M., & Forsyth, D. M. (2021). *The Handbook of New Zealand Mammals* (Third edition). Csiro Publishing. https://www.otago.ac.nz/press/books/otago756055.html

King, J. R., & Tschinkel, W. R. (2008). Experimental evidence that human impacts drive fire ant invasions and ecological change. *Proceedings of the National Academy of Sciences, 105*(51), 20339–20343. https://doi.org/10.1073/pnas.0809423105

Kitada, S. (2018). Economic, ecological and genetic impacts of marine stock enhancement and sea ranching: A systematic review. *Fish and Fisheries, 19*(3), 511–532. https://doi.org/10.1111/faf.12271

Kitajima, K., Fox, A. M., Sato, T., & Nagamatsu, D. (2006). Cultivar selection prior to introduction may increase invasiveness: Evidence from *Ardisia crenata*. *Biological Invasions, 8*(7), 1471. https://doi.org/10.1007/s10530-005-5839-9

Klein, J., Ruitton, S., Verlaque, M., & Boudouresque, C. (2005). Species introductions, diversity and disturbances in marine macrophyte assemblages of the northwestern Mediterranean Sea. *Marine Ecology Progress Series, 290*, 79–88. https://doi.org/10.3354/meps290079

Klein, R. A. (2011). Responsible Cruise Tourism: Issues of Cruise Tourism and Sustainability. *Journal of Hospitality and Tourism Management, 18*(1), 107–116. https://doi.org/10.1375/jhtm.18.1.107

Kliejunas, J. T. (2011). A risk assessment of climate change and the impact of forest diseases on forest ecosystems in the Western United States and Canada. *General Technical Report PSW-GTR-236. Albany, CA: USDA-Forest Service, Pacific Southwest Research Station. 70 p., 236*, 1–70. https://doi.org/10.2737/PSW-GTR-236

Knevel, I. C., Lans, T., Menting, F. B. J., Hertling, U. M., & van der Putten, W. H. (2004). Release from native root herbivores and biotic resistance by soil pathogens in a new habitat both affect the alien *Ammophila arenaria* in South Africa. *Oecologia, 141*(3), 502–510. https://doi.org/10.1007/s00442-004-1662-8

Kohli, R. K., Batish, D. R., Singh, H. P., & Dogra, K. S. (2006). Status, invasiveness and environmental threats of three tropical American invasive weeds (*Parthenium hysterophorus* L., *Ageratum conyzoides* L., *Lantana camara* L.) in India. *Biological Invasions, 8*(7), 1501–1510. https://doi.org/10.1007/s10530-005-5842-1

Kohli, R. K., Batish, D. R., Singh, J. S., Singh, H. P., & Bhatt, J. R. (2011). Plant invasion in India: An overview. In *Invasive alien plants: An ecological appraisal for the Indian subcontinent* (pp. 1–9). CABI. https://doi.org/10.1079/9781845939076.0001

Kolar, C. S., & Lodge, D. M. (2001). Progress in invasion biology: Predicting invaders. *Trends in Ecology & Evolution, 16*(4), 199–204. https://doi.org/10.1016/S0169-5347(01)02101-2

Kornis, M. S., Mercado-Silva, N., & Vander Zanden, M. J. (2012). Twenty years of invasion: A review of round goby *Neogobius melanostomus* biology, spread and ecological implications. *Journal of Fish Biology, 80*(2), 235–285. https://doi.org/10.1111/j.1095-8649.2011.03157.x

Kosaka, Y., Saikia, B., Mingki, T., Tag, H., Riba, T., & Ando, K. (2010). Roadside Distribution Patterns of Invasive Alien Plants Along an Altitudinal Gradient in Arunachal Himalaya, India. *Mountain Research and Development, 30*(3), 252–258. https://doi.org/10.1659/MRD-JOURNAL-D-10-00036.1

Kota, N. L., Landenberger, R. E., & McGraw, J. B. (2007). Germination and early growth of *Ailanthus* and tulip poplar in three levels of forest disturbance. *Biological Invasions, 9*(2), 197–211. https://doi.org/10.1007/s10530-006-9026-4

Krawczyk, R., Zubel, R., & Komsta, Ł. (2019). Military Training Areas and Vegetation – the Effect of Explosion Craters on Species Diversity along a Moisture Gradient. *Polish Journal of Ecology, 67*(3), 194–205. https://doi.org/10.3161/15052249PJE2019.67.3.003

Krkošek, M., Lewis, M. A., & Volpe, J. P. (2005). Transmission dynamics of parasitic sea lice from farm to wild salmon. Proceedings. *Biological Sciences, 272*(1564), 689–696. https://doi.org/10.1098/rspb.2004.3027

Krysko, K. L., Somma, L. A., Smith, D. C., Gillette, C. R., Cueva, D., Wasilewski, J. A., Enge, K. M., Johnson, S. A., Campbell, T. S., Edwards, J. R., Rochford, M. R., Tompkins, R., Fobb, J. L., Mullin, S., Lechowicz, C., Hazelton, D., & Warren, A. (2016). New verified nonindigenous amphibians and reptiles in Florida, 1976 through 2015, with a summary of over 152 years of introductions. *Reptiles & Amphibians, 23*(2), 110–143. https://doi.org/10.17161/randa.v23i2.14119

Kueffer, C., McDougall, K., Alexander, J., Daehler, C., Edwards, P., Haider, S., Milbau, A., Parks, C., Pauchard, A., Reshi, Z. A., Rew, L. J., Schroder, M., & Seipel, T. (2013). Plant Invasions into Mountain Protected Areas: Assessment, Prevention and Control at Multiple Spatial Scales. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7, pp. 89–113). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_6

Kühn, I., Wolf, J., & Schneider, A. (2017). Is there an urban effect in alien plant invasions? *Biological Invasions, 19*(12), 3505–3513. https://doi.org/10.1007/s10530-017-1591-1

Kummu, M., Moel, H. de, Salvucci, G., Viviroli, D., Ward, P. J., & Varis, O. (2016). Over the hills and further away from coast: Global geospatial patterns of human and environment over the 20th–21st centuries. *Environmental Research Letters, 11*(3), 034010. https://doi.org/10.1088/1748-9326/11/3/034010

Ladd, B. (2016). Nitrogen Pollution and the Meltdown of Urban Ecosystems. *Land, 5*(3), Article 3. https://doi.org/10.3390/land5030023

Lake, P. S., & O'Dowd, D. J. (1991). Red Crabs in Rain Forest, Christmas Island: Biotic Resistance to Invasion by an Exotic Snail. *Oikos, 62*(1), 25–29. https://doi.org/10.2307/3545442

Lambdon, P. W., Pyšek, P., Başnou, C., Hejda, M., Arianoutsou, M., Essl, F., Jarošík, V., Pergl, J., Winter, M., Anastasiu, P., Andriopoulos, P., Bazos, I., Brundu, G., Celesti-Grapow, L., Chassot, P., Delipetrou, P., Josefsson, M., Kark, S., Klotz, S., … Hulme, P. E. (2008). Alien flora of Europe: Species diversity, temporal trends, geographical patterns and research needs. *Preslia, 80*(2), 101–149. http://www.preslia.cz/P082Lam.pdf

Langor, D. W., Cameron, E. K., MacQuarrie, C. J. K., McBeath, A., McClay, A., Peter, B., Pybus, M., Ramsfield, T., Ryall, K., Scarr, T., Yemshanov, D., DeMerchant, I., Foottit, R., & Pohl, G. R. (2014). Non-native species in Canada's boreal zone: Diversity, impacts, and risk. *Environmental Reviews, 22*(4), 372–420. https://doi.org/10.1139/er-2013-0083

Laurance, W. F., & Peres, C. A. (2006). *Emerging threats to tropical forests.* University of Chicago Press.

Lawer, E. A., Mupepele, A.-C., & Klein, A.-M. (2019). Responses of small mammals to land restoration after mining. *Landscape Ecology, 34*(3), 473–485. https://doi.org/10.1007/s10980-019-00785-z

Lázaro-Lobo, A., & Ervin, G. N. (2019). A global examination on the differential impacts of roadsides on native vs. Exotic and weedy plant species. *Global Ecology and Conservation, 17*, e00555. https://doi.org/10.1016/j.gecco.2019.e00555

Le Maitre, D. C., Gaertner, M., Marchante, E., Ens, E.-J., Holmes, P. M., Pauchard, A., O'Farrell, P. J., Rogers, A. M., Blanchard, R., Blignaut, J., & Richardson, D. M. (2011). Impacts of invasive Australian *acacias*: Implications for management and restoration: Australian *acacias*: linking impacts and restoration. *Diversity and Distributions, 17*(5), 1015–1029. https://doi.org/10.1111/j.1472-4642.2011.00816.x

Le Preste, P. (Ed.). (2017). *Governing Global Biodiversity: The Evolution and Implementation of the Convention on Biological Diversity.* Routledge.

Leakey, A. D. B. (2009). Rising atmospheric carbon dioxide concentration and the future of C4 crops for food and fuel. *Proceedings of the Royal Society B: Biological Sciences, 276*(1666), Article 1666. https://doi.org/10.1098/rspb.2008.1517

Leclerc, C., Courchamp, F., & Bellard, C. (2018). Insular threat associations within taxa worldwide. *Scientific Reports, 8*(1), 6393. https://doi.org/10.1038/s41598-018-24733-0

Lee, J., Terauds, A., & Chown, S. (2014). Natural dispersal to sub-Antarctic Marion Island of two arthropod species. *Polar Biology, 37*(6), 781–787. https://doi.org/10.1007/s00300-014-1479-9

Lees, A. C., & Bell, D. J. (2008). A conservation paradox for the 21st century: The European wild rabbit *Oryctolagus cuniculus*, an invasive alien and an endangered native species. *Mammal Review, 38*(4), Article 4. https://doi.org/10.1111/j.1365-2907.2008.00116.x

Lehan, N. E., Murphy, J. R., Thorburn, L. P., & Bradley, B. A. (2013). Accidental introductions are an important source of invasive plants in the continental United States. *American Journal of Botany, 100*(7), 1287–1293. https://doi.org/10.3732/ajb.1300061

Lehmann, P., Ammunét, T., Barton, M., Battisti, A., Eigenbrode, S. D., Jepsen, J. U., Kalinkat, G., Neuvonen, S., Niemelä, P., & Terblanche, J. S. (2020). Complex responses of global insect pests to climate warming. *Frontiers in Ecology and the Environment, 18*(3), 141–150. https://doi.org/10.1002/fee.2160

Leishman, M. R., & Thomson, V. P. (2005). Experimental evidence for the effects of additional water, nutrients and physical disturbance on invasive plants in low fertility Hawkesbury Sandstone soils, Sydney, Australia. *Journal of Ecology, 93*(1), 38–49. https://doi.org/10.1111/j.1365-2745.2004.00938.x

Lejeusne, C., Chevaldonné, P., Pergent-Martini, C., Boudouresque, C. F., & Pérez, T. (2010). Climate change effects on a miniature ocean: The highly diverse, highly impacted Mediterranean Sea. *Trends in Ecology & Evolution, 25*(4), 250–260. https://doi.org/10.1016/j.tree.2009.10.009

Lembrechts, J. J., Alexander, J. M., Cavieres, L. A., Haider, S., Lenoir, J., Kueffer, C., McDougall, K., Naylor, B. J., Nuñez, M. A., Pauchard, A., Rew, L. J., Nijs, I., & Milbau, A. (2017). Mountain roads shift native and non-native plant species' ranges. *Ecography, 40*(3), 353–364. https://doi.org/10.1111/ecog.02200

Lembrechts, J. J., Pauchard, A., Lenoir, J., Nuñez, M. A., Geron, C., Ven, A., Bravo-Monasterio, P., Teneb, E., Nijs, I., & Milbau, A. (2016). Disturbance is the key to plant invasions in cold environments. *Proceedings of the National Academy of Sciences, 113*(49), 14061–14066. https://doi.org/10.1073/pnas.1608980113

Lemke, D., Schweitzer, C. J., Tadesse, W., Wang, Y., & Brown, J. A. (2013). Geospatial Assessment of Invasive Plants on Reclaimed Mines in Alabama. *Invasive Plant Science and Management, 6*(3), 401–410. https://doi.org/10.1614/IPSM-D-12-00045.1

Lenda, M., Skórka, P., Knops, J. M. H., Moroń, D., Sutherland, W. J., Kuszewska, K., & Woyciechowski, M. (2014). Effect of the Internet Commerce on Dispersal Modes of Invasive Alien Species. *PLoS ONE, 9*(6), e99786. https://doi.org/10.1371/journal.pone.0099786

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., & Geschke, A. (2012). International trade drives biodiversity threats in developing nations. *Nature, 486*(7401), 109–112. https://doi.org/10.1038/nature11145

Lenzner, B., Leclère, D., Franklin, O., Seebens, H., Roura-Pascual, N., Obersteiner, M., Dullinger, S., & Essl, F. (2019). A Framework for Global Twenty-First Century Scenarios and Models of Biological Invasions. *BioScience, 69*(9), 697–710. https://doi.org/10.1093/biosci/biz070

Leung, B., Roura-Pascual, N., Bacher, S., Heikkila, J., Brotons, L., Burgman, M. A., Dehnen-Schmutz, K., Essl, F., Hulme, P. E., Richardson, D. M., Sol, D., & Vilà, M. (2012). TEASIng apart alien species risk assessments: A framework for best practices. *Ecology Letters, 15*(12), 1475–1493. https://doi.org/10.1111/ele.12003

Leuven, R. S. E. W., van der Velde, G., Baijens, I., Snijders, J., van der Zwart, C., Lenders, H. J. R., & bij de Vaate, A. (2009). The river Rhine: A global highway for dispersal of aquatic invasive species. *Biological Invasions, 11*(9), 1989–2008. https://doi.org/10.1007/s10530-009-9491-7

Levin, P. S., Coyer, J. A., Petrik, R., & Good, T. P. (2002). Community-wide effects of nonindigenous species on temperate rocky reefs. *Ecology, 83*(11), 3182–3193. https://doi.org/10.2307/3071852

Levine, J. M., Adler, P. B., & Yelenik, S. G. (2004). A meta-analysis of biotic resistance to exotic plant invasions. *Ecology Letters, 7*(10), 975–989. https://doi.org/10.1111/j.1461-0248.2004.00657.x

Levine, J. M., & D'Antonio, C. M. (2003). Forecasting Biological Invasions with Increasing International Trade. *Conservation Biology, 17*(1), 322–326. https://doi.org/10.1046/j.1523-1739.2003.02038.x

Lewis, S. C., Blake, S. A., Trewin, B., Black, M. T., Dowdy, A. J., Perkins-Kirkpatrick, S. E., King, A. D., & Sharples, J. J. (2019). Deconstructing factors contributing to the 2018 fire weather in Queensland, Australia. *Bulletin of the American Meteorological Society, 10*(1), S115–S122. https://www.jstor.org/stable/27032754

Licata, F., Ficetola, G. F., Freeman, K., Mahasoa, R. H., Ravololonarivo, V., Solofo Niaina Fidy, J. F., Koto-Jean, A. B., Nahavitatsara, E. R., Andreone, F., & Crottini, A. (2019). Abundance, distribution and spread of the invasive Asian toad *Duttaphrynus melanostictus* in eastern Madagascar. *Biological Invasions, 21*(5), 1615–1626. https://doi.org/10.1007/s10530-019-01920-2

Liebhold, A. M., Brockerhoff, E. G., Kalisz, S., Nuñez, M. A., Wardle, D. A., & Wingfield, M. J. (2017). Biological invasions in forest ecosystems. *Biological Invasions, 19*(11), 3437–3458. https://doi.org/10.1007/s10530-017-1458-5

Liedtke, R., Barros, A., Essl, F., Lembrechts, J. J., Wedegärtner, R. E. M., Pauchard, A., & Dullinger, S. (2020). Hiking trails as conduits for the spread of non-native species in mountain areas. *Biological Invasions, 22*(3), 1121–1134. https://doi.org/10.1007/s10530-019-02165-9

Lieurance, D., Cooper, A., Young, A. L., Gordon, D. R., & Flory, S. L. (2018). Running bamboo species pose a greater invasion risk than clumping bamboo species in the continental United States. *Journal for Nature Conservation, 43*, 39–45. https://doi.org/10.1016/j.jnc.2018.02.012

Lin, Y., Gao, Z., & Zhan, A. (2015). Introduction and use of non-native species for aquaculture in China: Status, risks and management solutions. *Reviews in Aquaculture, 7*(1), 28–58. https://doi.org/10.1111/raq.12052

Lindenmayer, D. B., Hulvey, K. B., Hobbs, R. J., Colyvan, M., Felton, A., Possingham, H., Steffen, W., Wilson, K., Youngentob, K., & Gibbons, P. (2012). Avoiding bio-perversity from carbon sequestration solutions. *Conservation Letters, 5*(1), 28–36. https://doi.org/10.1111/j.1755-263X.2011.00213.x

Linders, T. E., Bekele, K., Schaffner, U., Allan, E., Alamirew, T., Choge, S. K., Haji, J., Muturi, G., Mbaabu, P. R., Shiferaw, H., & Eschen, R. (2020). The impact of invasive species on social-ecological systems: Relating supply and use of selected provisioning ecosystem services. *Ecosystem Services, 41*, 101055. https://doi.org/10.1016/j.ecoser.2019.101055

Lintermans, M. (2004). Human-assisted dispersal of alien freshwater fish in Australia. *New Zealand Journal of Marine and Freshwater Research, 38*(3), 481–501. https://doi.org/10.1080/00288330.2004.9517255

Lisi, P. J., Childress, E. S., Gagne, R. B., Hain, E. F., Lamphere, B. A., Walter, R. P., Hogan, J. D., Gilliam, J. F., Blum, M. J., & McIntyre, P. B. (2018). Overcoming urban stream syndrome: Trophic flexibility confers resilience in a Hawaiian stream fish. *Freshwater Biology, 63*(5), 492–502. https://doi.org/10.1111/fwb.13091

Litchman, E. (2010). Invisible invaders: Non-pathogenic invasive microbes in aquatic and terrestrial ecosystems. *Ecology Letters, 13*(12), 1560–1572. https://doi.org/10.1111/j.1461-0248.2010.01544.x

Liu, D., Wang, R., Gordon, D. R., Sun, X., Chen, L., & Wang, Y. (2017). Predicting Plant Invasions Following China's Water Diversion Project. *Environmental Science & Technology, 51*(3), 1450–1457. https://doi.org/10.1021/acs.est.6b05577

Liu, J., & Li, Z. (2010). The Role of Exotics in Chinese Inland Aquaculture. In S. S. De Silva & F. B. Davy (Eds.), *Success Stories in Asian Aquaculture* (pp. 173–185). Springer. https://doi.org/10.1007/978-90-481-3087-0_9

Livingstone, P., Hancox, N., Nugent, G., & de Lisle, G. (2015). Toward eradication: The effect of *Mycobacterium bovis* infection in wildlife on the evolution and future direction of bovine tuberculosis management in New Zealand. *New Zealand Veterinary Journal, 63*(sup1), Article sup1. https://doi.org/10.1080/00480169.2014.971082

Loarie, S. R., Duffy, P. B., Hamilton, H., Asner, G. P., Field, C. B., & Ackerly, D. D. (2009). The velocity of climate change. *Nature, 462*(7276), 1052–1055. https://doi.org/10.1038/nature08649

Lobo, D., Leao, T., Melo, F. P. L., Santos, A. M. M., & Tabarelli, M. (2011). Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. *Diversity and Distributions, 17*(2), 287–296. https://doi.org/10.1111/j.1472-4642.2010.00739.x

Lodge, D. M., Deines, A., Gherardi, F., Yeo, D. C. J., Arcella, T., Baldridge, A. K., Barnes, M. A., Chadderton, W. L., Feder, J. L., Gantz, C. A., Howard, G. W., Jerde, C. L., Peters, B. W., Peters, J. A., Sargent, L. W., Turner, C. R., Wittmann, M. E., & Zeng, Y. (2012). Global introductions of crayfishes: Evaluating the impact of species invasions on ecosystem services. *Annual Review of Ecology, Evolution, and Systematics, 43*(1), 449–472. https://doi.org/10.1146/annurev-ecolsys-111511-103919

Long, J. L. (2003). *Introduced Mammals of the World: Their History, Distribution and Influence.* CABI Publishing. https://doi.org/10.1071/9780643090156

Loomans, A. J., Vierbergen, B., & Chen, P.-P. (2013). Seven years of post-release monitoring for biological control agents released for biological control in the Netherlands: Impact of regulatory measures on establishment and use. *IOBC-WPRS Bulletin, 94*, 15–20.

Lorentz, K. A., & Minogue, P. J. (2015). Exotic *Eucalyptus* plantations in the southeastern US: risk assessment, management and policy approaches. *Biological Invasions, 17*(6), Article 6. https://doi.org/10.1007/s10530-015-0844-0

Lotze, H. K., Lenihan, H. S., Bourque, B. J., Bradbury, R. H., Cooke, R. G., Kay, M. C., Kidwell, S. M., Kirby, M. X., Peterson, C. H., & Jackson, J. B. C. (2006). Depletion, Degradation, and Recovery Potential of Estuaries and Coastal Seas. *Science, 312*(5781), 1806–1809. https://doi.org/10.1126/science.1128035

Lovette, I., Seutin, G., Ricklefs, R., & Birmingham, E. (1999). The assembly of an island fauna by natural invasion: Sources and temporal patterns in the avian colonization of Barbados. *Biological Invasions, 1*(1), 33–41. https://doi.org/doi.org/10.1023/A:1010090414598

Lowe, S., Browne, M., Boudjelas, S., & De Poorter, M. (2000). *100 of the world's worst invasive alien species: A selection from the global invasive species database.* Invasive Species Specialist Group Auckland, New Zealand. www.issg.org/booklet.pdf

Lozon, J. D., & MacIsaac, H. J. (1997). Biological invasions: Are they dependent on disturbance? *Environmental Reviews, 5*(2), 131–144. https://doi.org/10.1139/a97-007

Luce, C., Morgan, P., Dwire, K., Isaak, D., Holden, Z., Rieman, B., Gresswell, R., Rinne, J., Neville, H., Gresswell, R., Dunham, J., Rosenberger, A., & Young, M. (2012). Climate Change, Forests, Fire, Water, and Fish: Building resilient landscapes, streams, and managers. *JFSP Synthesis Reports.* https://digitalcommons.unl.edu/jfspsynthesis/2

Luláková, P., Perez-Mon, C., Šantrůčková, H., Ruethi, J., & Frey, B. (2019). High-Alpine Permafrost and Active-Layer Soil Microbiomes Differ in Their Response to Elevated Temperatures. *Frontiers in Microbiology, 10*, 668. https://doi.org/10.3389/fmicb.2019.00668

Lunt, I. D., Byrne, M., Hellmann, J. J., Mitchell, N. J., Garnett, S. T., Hayward, M. W., Martin, T. G., McDonald-Maddden, E., Williams, S. E., & Zander, K. K. (2013). Using assisted colonisation to conserve biodiversity and restore ecosystem function under climate change. *Biological Conservation, 157*, 172–177. https://doi.org/10.1016/j.biocon.2012.08.034

Luo, D., Wei, H., Chaichana, R., Yang, D., Gu, D., Mu, X., Xu, M., Yang, Y., Jin, S., & Hu, Y. (2019). Current status and potential risks of established alien fish species in China. *Aquatic Ecosystem Health & Management, 22*(4), 371–384. https://doi.org/10.1080/14634988.2019.1693226

Luomba, J. O. (2016). *Illegal, unreported and unregulated (IUU) fishing as a governability problem: A case study of Lake Victoria, Tanzania* [Masters, Memorial University of Newfoundland]. https://research.library.mun.ca/12424/

Macdonald, S. E., Landhäusser, S. M., Skousen, J., Franklin, J., Frouz, J., Hall, S., Jacobs, D. F., & Quideau, S. (2015). Forest restoration following surface mining disturbance: Challenges and solutions. *New Forests, 46*(5), 703–732. https://doi.org/10.1007/s11056-015-9506-4

Maceda-Veiga, A., Escribano-Alacid, J., de Sostoa, A., & Garcia-Berthou, E. (2013). The aquarium trade as a potential source of fish introductions in southwestern Europe. *Biological Invasions, 15*(12), 2707–2716. https://doi.org/10.1007/s10530-013-0485-0

Madzivanzira, T. C., South, J., Wood, L. E., Nunes, A. L., & Weyl, O. L. F. (2021). A Review of Freshwater Crayfish Introductions in Africa. *Reviews in Fisheries Science & Aquaculture, 29*(2), 218–241. https://doi.org/10.1080/23308249.2020.1802405

Magalhães, A. L. B. de, & Jacobi, C. M. (2010). E-commerce of freshwater aquarium fishes: Potential disseminator of exotic species in Brazil. *Acta Scientiarum. Biological Sciences, 32*(3), 243–248. https://doi.org/10.4025/actascibiolsci.v32i3.3919

Malumbres-Olarte, J., Barratt, B. I., Vink, C. J., Paterson, A. M., Cruickshank, R. H., Ferguson, C. M., & Barton, D. M. (2014). Big and aerial invaders: Dominance of exotic spiders in burned New Zealand tussock grasslands. *Biological Invasions, 16*(11), 2311–2322. https://doi.org/10.1007/s10530-014-0666-5

Mandle, L., Bufford, J. L., Schmidt, I. B., & Daehler, C. C. (2011). Woody exotic plant invasions and fire: Reciprocal impacts and consequences for native ecosystems. *Biological Invasions, 13*(8), 1815–1827. https://doi.org/10.1007/s10530-011-0001-3

Mandon-Dalger, I., Clergeau, P., Tassin, J., Rivière, J.-N., & Gatti, S. (2004). Relationships between Alien Plants and an Alien Bird Species on Reunion Island. *Journal of Tropical Ecology, 20*(6), 635–642. https://www.jstor.org/stable/4092109

Manzoor, S. A., Griffiths, G., & Lukac, M. (2021). Land use and climate change interaction triggers contrasting trajectories of biological invasion. *Ecological Indicators, 120*, 106936. https://doi.org/10.1016/j.ecolind.2020.106936

Marbuah, G., Gren, I.-M., & McKie, B. (2014). Economics of Harmful Invasive Species: A Review. *Diversity, 6*(3), 500–523. https://doi.org/10.3390/d6030500

Marín-Guirao, L., Bernardeau-Esteller, J., Ruiz, J. M., & Sandoval-Gil, J. M. (2015). Resistance of *Posidonia oceanica* seagrass meadows to the spread of the introduced green alga *Caulerpa cylindracea*: Assessment of the role of light. *Biological Invasions, 17*(7), 1989–2009. https://doi.org/10.1007/s10530-015-0852-0

Marini, L., Battisti, A., Bona, E., Federici, G., Martini, F., Pautasso, M., & Hulme, P. E. (2012). Alien and native plant life-forms respond differently to human and climate pressures. *Global Ecology and Biogeography, 21*(5), 534–544. https://doi.org/10.1111/j.1466-8238.2011.00702.x

Marler, T. E., & Krishnapillai, M. V. (2020). Longitude, Forest Fragmentation, and Plant Size Influence *Cycas micronesica* Mortality Following Island Insect Invasions. *Diversity, 12*(5), 194. https://doi.org/10.3390/d12050194

Maron, J., & Marler, M. (2007). Native plant diversity resists invasion at both low and high resource levels. *Ecology, 88*(10), 2651–2661. https://doi.org/10.1890/06-1993.1

Marrack, L. (2016). Modeling Potential Shifts in Hawaiian Anchialine Pool Habitat and Introduced Fish Distribution due to Sea Level Rise. *Estuaries and Coasts, 39*(3), 781–797. https://doi.org/10.1007/s12237-015-0025-5

Marshall, B. E. (2018). Guilty as charged: Nile perch was the cause of the haplochromine decline in Lake Victoria. *Canadian Journal of Fisheries and Aquatic Sciences, 75*(9), 1542–1559. https://doi.org/10.1139/cjfas-2017-0056

Marshall, V. M., Lewis, M. M., & Ostendorf, B. (2012). Buffel grass *(Cenchrus ciliaris)* as an invader and threat to biodiversity in arid environments: A review. *Journal of Arid Environments, 78*, 1–12. https://doi.org/10.1016/j.jaridenv.2011.11.005

Martin, T. G., Murphy, H., Liedloff, A., Thomas, C., Chadès, I., Cook, G., Fensham, R., McIvor, J., & van Klinken, R. D. (2015). Buffel grass and climate change: A framework for projecting invasive species distributions when data are scarce. *Biological Invasions, 17*(11), 3197–3210. https://doi.org/10.1007/s10530-015-0945-9

Martinez-Cordero, F. J., & Sanchez-Zazueta, E. (2022). Poverty and vulnerability assessment of tilapia farmers in the southwestern States of Oaxaca, Guerrero, and Chiapas in Mexico. *Aquaculture Economics & Management, 26*(1), 36–56. https://doi.org/10.1080/13657305.2021.1896604

Mason, S. C., Palmer, G., Fox, R., Gillings, S., Hill, J. K., Thomas, C. D., & Oliver, T. H. (2015). Geographical range margins of many taxonomic groups continue to shift polewards. *Biological Journal of the Linnean Society, 115*(3), 586–597. https://doi.org/10.1111/bij.12574

Massa, C., Doyle, M., & Fortunato, R. C. (2014). On how Cattle Egret *(Bubulcus ibis)* spread to the Americas: Meteorological tools to assess probable colonization trajectories. *International Journal of Biometeorology, 58*, 1879–1891. https://doi.org/10.1007/s00484-014-0790-z

Masters, G., & Norgrove, L. (2010). *Climate change and invasive alien species* (p. 30) [CABI Working Paper 1]. https://www.cabi.org/Uploads/CABI/expertise/invasive-alien-species-working-paper.pdf

Maus, V., Giljum, S., Gutschlhofer, J., da Silva, D. M., Probst, M., Gass, S. L. B., Luckeneder, S., Lieber, M., & McCallum, I. (2020). A global-scale data set of mining areas. *Scientific Data, 7*(1), Article 1. https://doi.org/10.1038/s41597-020-00624-w

Mayer, K., Haeuser, E., Dawson, W., Essl, F., Kreft, H., Pergl, J., Pyšek, P., Weigelt, P., Winter, M., Lenzner, B., & van Kleunen, M. (2017). Naturalization of ornamental plant species in public green spaces and private gardens. *Biological Invasions, 19*(12), Article 12. https://doi.org/10.1007/s10530-017-1594-y

Mayonde, S. G., Cron, G. V., Gaskin, J. F., & Byrne, M. J. (2015). Evidence of *Tamarix* hybrids in South Africa, as inferred by nuclear ITS and plastid *trn*S–*trn*G DNA sequences. *South African Journal of Botany, 96*, 122–131. https://doi.org/10.1016/j.sajb.2014.10.011

Mazur, M. C., Kowalski, K., & David Galbraith, D. (2014). Assessment of suitable habitat for *Phragmites australis* (common reed) in the Great Lakes coastal zone. *Aquatic Invasions, 9*(1), 1–19. https://doi.org/10.3391/ai.2014.9.1.01

Mazza, G., Aquiloni, L., Inghilesi, A. F., Giuliani, C., Lazzaro, L., Ferretti, G., Lastrucci, L., Foggi, B., & Tricarico, E. (2015). Aliens just a click away: The online aquarium trade in Italy. *Management of Biological Invasions, 6*(3), 253–261. https://doi.org/10.3391/mbi.2015.6.3.04

Mazzamuto, M. V., Wauters, L. A., & Koprowski, J. L. (2021). Exotic Pet Trade as a Cause of Biological Invasions: The Case of Tree Squirrels of the Genus *Callosciurus. Biology, 10*(10), 1046. https://doi.org/10.3390/biology10101046

McCullough, D. G., Work, T. T., Cavey, J. F., Liebhold, A. M., & Marshall, D. (2006). Interceptions of nonindigenous plant pests at US ports of entry and border crossings over a 17-year period. *Biological Invasions, 8*(4), 611–630. https://doi.org/10.1007/s10530-005-1798-4

McDougall, K. L., Alexander, J. M., Haider, S., Pauchard, A., Walsh, N. G., & Kueffer, C. (2011). Alien flora of mountains: Global comparisons for the development of local preventive measures against plant invasions. *Diversity and Distributions, 17*(1), 103–111. https://doi.org/10.1111/j.1472-4642.2010.00713.x

McGregor, H. W., Legge, S., Jones, M. E., & Johnson, C. N. (2014). Landscape Management of Fire and Grazing Regimes Alters the Fine-Scale Habitat Utilisation by Feral Cats. *PLoS ONE, 9*(10), e109097. https://doi.org/10.1371/journal.pone.0109097

McGregor, H. W., Legge, S., Jones, M. E., & Johnson, C. N. (2016). Extraterritorial hunting expeditions to intense fire scars by feral cats. *Scientific Reports, 6*(1), Article 1. https://doi.org/10.1038/srep22559

McGregor, K. F., Watt, M. S., Hulme, P. E., & Duncan, R. P. (2012). What determines pine naturalization: Species traits, climate suitability or forestry use? *Diversity and Distributions, 18*(10), 1013–1023. https://doi.org/10.1111/j.1472-4642.2012.00942.x

McKenzie, L. A., Brooks, R. C., & Johnston, E. L. (2012). A widespread contaminant enhances invasion success of a marine invader. *Journal of Applied Ecology, 49*(4), 767–773. https://doi.org/10.1111/j.1365-2664.2012.02158.x

McKinney, M. L. (2002). Do human activities raise species richness? Contrasting patterns in United States plants and fishes. *Global Ecology and Biogeography, 11*(4), 343–348. https://doi.org/10.1046/j.1466-822X.2002.00293.x

McNeely, J. A. (2001). An introduction to human dimensions of invasive alien species. In *The Great Reshuffling; Human Dimensions of Invasive Alien Species* (pp. 1–9). IUCN – The World Conservation Union.

McQuaid, C. D., & Phillips, T. E. (2000). Limited wind-driven dispersal of intertidal mussel larvae: *In situ* evidence from the plankton and the spread of the invasive species *Mytilus galloprovincialis* in South Africa. *Marine Ecology Progress Series, 201*, 211–220. https://doi.org/10.3354/meps201211

Mdweshu, L., & Maroyi, A. (2020). Short Communication: Local perceptions about utilization of invasive alien species *Opuntia ficus-indica* in three Local Municipalities in the Eastern Cape Province, South Africa. *Biodiversitas Journal of Biological Diversity, 21*(4), Article 4. https://doi.org/10.13057/biodiv/d210446

Mead, A., Griffiths, C. L., Branch, G. M., McQuaid, C. D., Blamey, L. K., Bolton, J. J., Anderson, R. J., Dufois, F., Rouault, M., Froneman, P. W., Whitfield, A. K., Harris, L. R., Nel, R., Pillay, D., & Adams, J. B. (2013). Human-mediated drivers of change—Impacts on coastal ecosystems and marine biota of South Africa. *African Journal of Marine Science, 35*(3), 403–425. https://doi.org/10.2989/1814232X.2013.830147

Measey, J., Visser, V., Dgebuadze, Y., Inderjit, Li, B., Dechoum, M., Ziller, S. R., & Richardson, D. M. (2019). The world needs BRICS countries to build capacity in invasion science. *PLOS Biology, 17*(9), e3000404. https://doi.org/10.1371/journal.pbio.3000404

Medley, K. A., Jenkins, D. G., & Hoffman, E. A. (2015). Human-aided and natural dispersal drive gene flow across the range of an invasive mosquito. *Molecular Ecology*, 284–295. https://doi.org/10.1111/mec.12925

Megahed, M. E. (2014). Design breeding program for the Indian white shrimp *Fenneropenaeus indicus* in Egypt. Aquaculture America. https://www.researchgate.net/publication/310509811_DESIGN_BREEDING_PROGRAM_FOR_THE_INDIAN_WHITE_SHRIMP_Fenneropenaeus_indicus_IN_EGYPT

Meijer, J. R., Huijbregts, M. A. J., Schotten, K. C. G. J., & Schipper, A. M. (2018). Global patterns of current and future road infrastructure. *Environmental Research Letters, 13*(6), Article 6. https://doi.org/10.1088/1748-9326/aabd42

Migliorini, D., Ghelardini, L., Tondini, E., Luchi, N., & Santini, A. (2015). The potential of symptomless potted plants for carrying invasive soilborne plant pathogens. *Diversity and Distributions, 21*(10), 1218–1229. https://doi.org/10.1111/ddi.12347

Millennium Ecosystem Assessment (Ed.). (2005a). *Ecosystems and human well-being: Desertification synthesis.* Island Press.

Millennium Ecosystem Assessment. (2005b). *Ecosystems and Human Well-being: Synthesis*. Island Press.

Miller, G., Friedel, M., Adam, P., & Chewings, V. (2010). Ecological impacts of buffel grass (*Cenchrus ciliaris* L.) invasion in central Australia—Does field evidence support a fire-invasion feedback? *The Rangeland Journal, 32*(4), 353. https://doi.org/10.1071/RJ09076

Miller, J. A., Carlton, J. T., Chapman, J. W., Geller, J. B., & Ruiz, G. M. (2018). Transoceanic dispersal of the mussel *Mytilus galloprovincialis* on Japanese tsunami marine debris: An approach for evaluating rafting of a coastal species at sea. *Marine Pollution Bulletin, 132*, 60–69. https://doi.org/10.1016/j.marpolbul.2017.10.040

Mineur, F., Belsher, T., Johnson, M. P., Maggs, C. A., & Verlaque, M. (2007). Experimental assessment of oyster transfers as a vector for macroalgal introductions. *Biological Conservation, 137*(2), 237–247. https://doi.org/10.1016/j.biocon.2007.02.001

Mineur, F., Cook, E. J., Minchin, D., Bohn, K., MacLeod, A., & Maggs, C. A. (2012). Changing coasts: Marine aliens and artificial structures. In R. N. Gibson, R. J. A. Atkinson, J. D. M. Gordon, & R. N. Hughes (Eds.), *Oceanography and Marine Biology: An Annual Review* (First edition, Vol. 50, pp. 189–233). CRC Press. https://www.taylorfrancis.com/chapters/edit/10.1201/b12157-6/changing-coasts-marine-aliens-arti%C2%8Dcial-structures-fr%C3%A9d%C3%A9ric-mineur-elizabeth-cook-dan-minchin-katrin-bohn-adrian-macleod-christine--maggs?context=ubx&refId=2930e715-4984-4ce8-81b6-66fa5cbd65e9

Mineur, F., Le Roux, A., Maggs, C. A., & Verlaque, M. (2014). Positive feedback loop between introductions of non-native marine species and cultivation of oysters in Europe. *Conservation Biology: The Journal of the Society for Conservation Biology, 28*(6), 1667–1676. https://doi.org/10.1111/cobi.12363

Miralles, L., Gomez-Agenjo, M., Rayon-Viña, F., Gyraitė, G., & Garcia-Vazquez, E. (2018). Alert calling in port areas: Marine litter as possible secondary dispersal vector for hitchhiking invasive species. *Journal for Nature Conservation, 42, 12–18.* https://doi.org/10.1016/j.jnc.2018.01.005

Moles, A. T., Flores-Moreno, H., Bonser, S. P., Warton, D. I., Helm, A., Warman, L., Eldridge, D. J., Jurado, E., Hemmings, F. A., Reich, P. B., Cavender-Bares, J., Seabloom, E. W., Mayfield, M. M., Sheil, D., Djietror, J. C., Peri, P. L., Enrico, L., Cabido, M. R., Setterfield, S. A., … Thomson, F. J. (2012). Invasions: The trail behind, the path ahead, and a test of a disturbing idea. *Journal of Ecology, 100*(1), Article 1. https://doi.org/10.1111/j.1365-2745.2011.01915.x

Molina-Montenegro, M. A., Carrasco-Urra, F., Acuna-Rodriguez, I., Oses, R., Torres-Diaz, C., & Chwedorzewska, K. J. (2014). Assessing the importance of human activities for the establishment of the invasive *Poa annua* in Antarctica. *Polar Research, 33*, 21425. https://doi.org/10.3402/polar.v33.21425

Molina-Montenegro, M. A., Carrasco-Urra, F., Rodrigo, C., Convey, P., Valladares, F., & Gianoli, E. (2012). Occurrence of the Non-Native Annual Bluegrass on the Antarctic Mainland and Its Negative Effects on Native Plants. *Conservation Biology, 26*(4), 717–723. https://doi.org/10.1111/j.1523-1739.2012.01865.x

Molnar, J. L., Gamboa, R. L., Revenga, C., & Spalding, M. D. (2008). Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment, 6*(9), 485–492. https://doi.org/10.1890/070064

Montalva, J., Dudley, L., Arroyo, M. K., Retamales, H., & Abrahamovich, A. H. (2011). Geographic distribution and associated flora of native and introduced bumble bees (*Bombus* spp.) in Chile. *Journal of Apicultural Research, 50*(1), 11–21. https://doi.org/10.3896/IBRA.1.50.1.02

Montefalcone, M., Morri, C., Bianchi, C. N., Bavestrello, G., & Piazzi, L. (2017). The two facets of species sensitivity: Stress and disturbance on coralligenous assemblages in space and time. *Marine Pollution Bulletin, 117*(1–2), 229–238. https://doi.org/10.1016/j.marpolbul.2017.01.072

Moodley, D., Geerts, S., Richardson, D. M., & Wilson, J. R. U. (2013). Different Traits Determine Introduction, Naturalization and Invasion Success In Woody Plants: Proteaceae as a Test Case. *PLoS ONE, 8*(9), e75078. https://doi.org/10.1371/journal.pone.0075078

Moore, B. A. (2005). *Alien Invasive Species: Impacts on Forests and Forestry—A Review* (Working Paper 8; Forest Resources Development Service Working Paper FBS/8E). Forest Resources Division FAO. https://www.fao.org/3/j6854e/j6854e00.htm

Morales, C. L., & Aizen, M. A. (2002). Does Invasion of Exotic Plants Promote Invasion of Exotic Flower Visitors? A Case Study from the Temperate Forests of the Southern Andes. *Biological Invasions, 4*(1), 87–100. https://doi.org/10.1023/A:1020513012689

Morales, C. L., Arbetman, M. P., Cameron, S. A., & Aizen, M. A. (2013). Rapid ecological replacement of a native bumble bee by invasive species. *Frontiers in Ecology and the Environment, 11*(10), 529–534. https://doi.org/10.1890/120321

Morales, C. L., Sáez, A., Garibaldi, L. A., & Aizen, M. A. (2017). Disruption of Pollination Services by Invasive Pollinator Species. In M. Vilà & P. E. Hulme (Eds.), *Impact of Biological Invasions on Ecosystem Services* (Vol. 12, pp. 203–220). Springer International Publishing. https://doi.org/10.1007/978-3-319-45121-3_13

Moran, V. C., & Hoffmann, J. H. (2015). The fourteen International Symposia on Biological Control of Weeds, 1969–2014: Delegates, demographics and inferences from the debate on non-target effects. Biological Control, 87, 23–31. https://doi.org/10.1016/j.biocontrol.2015.04.008

Morrisey, D., Inglis, G., Neil, K., Bradley, A., & Fitridge, I. (2011). Characterization of the marine aquarium trade and management of associated marine pests in Australia, a country with stringent import biosecurity regulation. *Environmental Conservation, 38*(1), 89–100. https://doi.org/10.1017/S0376892911000014

Morrison, L. W., Porter, S. D., Daniels, E., & Korzukhin, M. D. (2004). Potential Global Range Expansion of the Invasive Fire Ant, *Solenopsis invicta. Biological Invasions, 6*(2), 183–191. https://doi.org/10.1023/B:BINV.0000022135.96042.90

Morse, N. B., Pellissier, P. A., Cianciola, E. N., Brereton, R. L., Sullivan, M. M., Shonka, N. K., Wheeler, T. B., & McDowell, W. H. (2014). Novel ecosystems in the Anthropocene: A revision of the novel ecosystem concept for pragmatic applications. *Ecology and Society, 19*(2), art12. https://doi.org/10.5751/ES-06192-190212

Mortensen, D. A., Rauschert, E. S. J., Nord, A. N., & Jones, B. P. (2009). Forest Roads Facilitate the Spread of Invasive Plants. *Invasive Plant Science and Management, 2*(3), 191–199. https://doi.org/10.1614/IPSM-08-125.1

Mosina, G. K. E., Maroyi, A., & Potgieter, M. J. (2015). Useful Plants Grown and Maintained in Domestic Gardens of the Capricorn District, Limpopo Province, South Africa. *Studies on Ethno-Medicine, 9*(1), 43–58. https://doi.org/10.1080/09735070.2015.11905420

Mosweu, S., Munyati, C., Kabanda, T., Setshogo, M., & Muzila, M. (2013). *Prosopis* L. Invasion in the South-Western Region of Botswana: The Perceptions of Rural Communities and Management Options. *Natural Resources, 4*(8), Article 8. https://doi.org/10.4236/nr.2013.48061

Mountain Research Initiative EDW Working Group. (2015). Elevation-dependent warming in mountain regions of the world. *Nature Climate Change, 5*(5), 424–430. https://doi.org/10.1038/nclimate2563

Mueller, J. M., & Hellmann, J. J. (2008). An assessment of invasion risk from assisted migration. *Conservation Biology, 22*(3), 562–567. https://doi.org/10.1111/j.1523-1739.2008.00952.x

Mueller, S. E., Thode, A. E., Margolis, E. Q., Yocom, L. L., Young, J. D., & Iniguez, J. M. (2020). Climate relationships with increasing wildfire in the southwestern US from 1984 to 2015. *Forest Ecology and Management, 460*, 117861. https://doi.org/10.1016/j.foreco.2019.117861

Mungi, N. A., Coops, N. C., Ramesh, K., & Rawat, G. S. (2018). How global climate change and regional disturbance can expand the invasion risk? Case study of *Lantana camara* invasion in the Himalaya. *Biological Invasions, 20*(7), 1849–1863. https://doi.org/10.1007/s10530-018-1666-7

Munoz, J., Felicisimo, Á. M., Cabezas, F., Burgaz, A. R., & Martinez, I. (2004). Wind as a long-distance dispersal vehicle in the Southern Hemisphere. *Science, 304*, 1144–1147. https://doi.org/10.1126/science.1095210

Murphy, S. T., & Cheesman, O. D. (2006). *The Aid Trade—International Assistance Programs as Pathways for the Introduction of Invasive Alien Species* (Biodiversity Series, p. 50) [A Preliminary Report]. World Bank Environmental Department.

Murray, C. C., Gartner, H., Gregr, E. J., Chan, K., Pakhomov, E., & Therriault, T. W. (2014). Spatial distribution of marine invasive species: Environmental, demographic and vector drivers. *Diversity and Distributions, 20*(7), 824–836. https://doi.org/10.1111/ddi.12215

Murray, J. V., Stokes, K. E., & Klinken, R. D. van. (2012). Predicting the potential distribution of a riparian invasive plant: The effects of changing climate, flood regimes and land-use patterns. *Global Change Biology, 18*(5), 1738–1753. https://doi.org/10.1111/j.1365-2486.2011.02621.x

Mwangi, E., & Swallow, B. (2008). *Prosopis juliflora* invasion and rural livelihoods in the Lake Baringo area of Kenya. *Conservation and Society, 6*(2), 130–140. https://doi.org/10.4103/0972-4923.49207

Myers, K., Parer, I., Wood, D., & Cooke, B. D. (1994). The rabbit in Australia. In H. Thompson & C. King (Eds.), *The European Rabbit: The History and Biology of a Successful Coloniser.* Oxford University Press: New York. https://edepot.wur.nl/469421#page=476

Nagarajan, S., & Singh, D. V. (1990). Long-distance dispersion of rust pathogens. *Annual Review of Phytopathology, 28,* 139–153. https://doi.org/10.1146/annurev.py.28.090190.001035

Navarro, L. A., & Phiri, G. (2000). *Water hyacinth in Africa and the Middle East: A survey of problems and solutions.*

Naylor, R. L., Williams, S. L., & Strong, D. R. (2001). Aquaculture—A Gateway for Exotic Species. *Science, 294*(5547), 1655–1656. https://doi.org/10.1126/science.1064875

Neill, P. E., & Arim, M. (2011). Human Health Link to Invasive Species. *Encyclopedia of Environmental Health,* 116–123. https://doi.org/10.1016/B978-0-444-52272-6.00528-6

Nel, P. J. (1983). *Deciduous Fruits and Vines: Pests and Diseases and Their Control.* David Philip, Cape Town, South Africa.

Nelson, G. C., Bennett, E., Berhe, A. A., Cassman, K., DeFries, R., Dietz, T., Dobermann, A., Dobson, A., Janetos, A., Levy, M., Marco, D., Nakicenovic, N., O'Neill, B., Norgaard, R., Petschel-Held, G., Ojima, D., Pingali, P., Watson, R., & Zurek, M. (2006). Anthropogenic drivers of ecosystem change: An overview. *Ecology and Society, 11*(2), 29. https://www.jstor.org/stable/26266018

Ni, M., & Hulme, P. E. (2021). Botanic gardens play key roles in the regional distribution of first records of alien plants in China. *Global Ecology and Biogeography, 30*(8), 1572–1582. https://doi.org/10.1111/geb.13319

Nicholls, R. J., Hanson, S. E., Lowe, J. A., Warrick, R. A., Lu, X., & Long, A. J. (2014). Sea-level scenarios for evaluating coastal impacts. *WIREs Climate Change, 5*(1), 129–150. https://doi.org/10.1002/wcc.253

Ning, Z., Xie, T., Liu, Z., Bai, J., & Cui, B. (2019). Native herbivores enhance the resistance of an anthropogenically disturbed salt marsh to *Spartina alterniflora* invasion. *Ecosphere, 10*(1). https://doi.org/10.1002/ecs2.2565

Nobis, M. P., Jaeger, J. A. G., & Zimmermann, N. E. (2009). Neophyte species richness at the landscape scale under urban sprawl and climate warming. *Diversity and Distributions, 15*(6), 928–939. https://doi.org/10.1111/j.1472-4642.2009.00610.x

Nolan, C., Overpeck, J. T., Allen, J. R., Anderson, P. M., Betancourt, J. L., Binney, H. A., Brewer, S., Bush, M. B., Chase, B. M., & Cheddadi, R. (2018). Past and future global transformation of terrestrial ecosystems under climate change. *Science, 361*(6405), 920–923. https://doi.org/10.1126/science.aan5360

Nolan, R. H., Boer, M. M., Collins, L., Resco de Dios, V., Clarke, H., Jenkins, M., Kenny, B., & Bradstock, R. A. (2020). Causes and consequences of eastern Australia's 2019–20 season of mega-fires. *Global Change Biology, 26*(3), 1039–1041. https://doi.org/10.1111/gcb.14987

Nowak, R. S., Ellsworth, D. S., & Smith, S. D. (2004). Functional responses of plants to elevated atmospheric CO_2—Do photosynthetic and productivity data from FACE experiments support early predictions? *New Phytologist, 162*(2), 253–280. https://doi.org/10.1111/j.1469-8137.2004.01033.x

Numbere, A. O. (2019). Impact of Invasive Nypa Palm *(Nypa Fruticans)* on Mangroves in Coastal Areas of the Niger Delta Region, Nigeria. In C. Makowski & C. W. Finkl (Eds.), *Impacts of Invasive Species on Coastal Environments: Coasts in Crisis* (pp. 425–454). Springer International Publishing. https://doi.org/10.1007/978-3-319-91382-7_13

Nunes, A. L., Tricarico, E., Panov, V. E., Cardoso, A. C., & Katsanevakis, S. (2015). Pathways and gateways of freshwater invasions in Europe. *Aquatic Invasions, 10*(4), 359–370. https://doi.org/10.3391/ai.2015.10.4.01

Nuñez, M. A., Chiuffo, M. C., Torres, A., Paul, T., Dimarco, R. D., Raal, P., Policelli, N., Moyano, J., García, R. A., van Wilgen, B. W., Pauchard, A., & Richardson, D. M. (2017). Ecology and management of invasive Pinaceae around the world: Progress and challenges. *Biological Invasions, 19*(11), 3099–3120. https://doi.org/10.1007/s10530-017-1483-4

Nuñez, M. A., Hayward, J., Horton, T. R., Amico, G. C., Dimarco, R. D., Barrios-Garcia, M. N., & Simberloff, D. (2013). Exotic Mammals Disperse Exotic Fungi That Promote Invasion by Exotic Trees. *PLoS ONE, 8*(6), e66832. https://doi.org/10.1371/journal.pone.0066832

Nwobi, C., Williams, M., & Mitchard, E. T. A. (2020). Rapid Mangrove Forest Loss and Nipa Palm *(Nypa fruticans)* Expansion in the Niger Delta, 2007–2017. *Remote Sensing, 12*(14), 2344. https://doi.org/10.3390/rs12142344

Obermeister, N. (2019). Local knowledge, global ambitions: IPBES and the advent of multi-scale models and scenarios. *Sustainability Science, 14*(3), 843–

856. https://doi.org/10.1007/s11625-018-0616-8

Ochoa-Hueso, R., Allen, E. B., Branquinho, C., Cruz, C., Dias, T., Fenn, M. E., Manrique, E., Pérez-Corona, M. E., Sheppard, L. J., & Stock, W. D. (2011). Nitrogen deposition effects on Mediterranean-type ecosystems: An ecological assessment. *Environmental Pollution, 159*(10), Article 10. https://doi.org/10.1016/j.envpol.2010.12.019

Ochoa-Hueso, R., & Manrique, E. (2010). Nitrogen Fertilization and Water Supply Affect Germination and Plant Establishment of the Soil Seed Bank Present in a Semi-Arid Mediterranean Scrubland. *Plant Ecology, 210*(2), 263–273. https://doi.org/10.1007/s11258-010-9755-4

O'Connor, T. G., & van Wilgen, B. W. (2020). The Impact of Invasive Alien Plants on Rangelands in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (pp. 459–487). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_16

Odom, R. L., Solomon, J. A., & Walters, L. J. (2014). Alternatives to Release: Efficient Methods for Disposal of Excess or Unwanted Aquarium Macroalgae in the Genus *Chaetomorpha. Invasive Plant Science and Management, 7*(1), 76–83. https://doi.org/10.1614/IPSM-D-13-00053.1

O'Donnell, J., Gallagher, R. V., Wilson, P. D., Downey, P. O., Hughes, L., & Leishman, M. R. (2012). Invasion hotspots for non-native plants in Australia under current and future climates. *Global Change Biology, 18*(2), 617–629. https://doi.org/10.1111/j.1365-2486.2011.02537.x

O'Dowd, D. J., Green, P. T., & Lake, P. S. (2003). Invasional 'meltdown' on an oceanic island. *Ecology Letters, 6*(9), 812–817. https://doi.org/10.1046/j.1461-0248.2003.00512.x

Ogutu-Ohwayo, R. (1998). Nile perch: The great lake experiment. *World Conservation, 1*, 11–12. https://pubmed.ncbi.nlm.nih.gov/12295947/

Ogutu-Ohwayo, R., Natugonza, V., Musinguzi, L., Olokotum, M., & Naigaga, S. (2016). Implications of climate variability and change for African lake ecosystems, fisheries productivity, and livelihoods. *Journal of Great Lakes Research, 42*(3), 498–510. https://doi.org/10.1016/j.jglr.2016.03.004

Oguz, T., Fach, B., & Salihoglu, B. (2008). Invasion dynamics of the alien ctenophore *Mnemiopsis leidyi* and its impact on anchovy collapse in the Black Sea. *Journal of Plankton Research, 30*(12), 1385–1397. https://doi.org/10.1093/plankt/fbn094

Ojaveer, H., Galil, B. S., Carlton, J. T., Alleway, H., Goulletquer, P., Lehtiniemi, M., Marchini, A., Miller, W., Occhipinti-Ambrogi, A., Peharda, M., Ruiz, G. M., Williams, S. L., & Zaiko, A. (2018). Historical baselines in marine bioinvasions: Implications for policy and management. *PLoS ONE, 13*(8), e0202383. https://doi.org/10.1371/journal.pone.0202383

Olfert, O., Weiss, R. M., & Elliott, R. H. (2016). Bioclimatic approach to assessing the potential impact of climate change on wheat midge (Diptera: Cecidomyiidae) in North America. *The Canadian Entomologist, 148*(1), 52–67. https://doi.org/10.4039/tce.2015.40

Olfert, O., Weiss, R. M., Elliott, R. H., & Soroka, J. J. (2017). Bioclimatic approach to assessing the potential impact of climate change on two flea beetle (Coleoptera: Chrysomelidae) species in Canada. *The Canadian Entomologist, 149*(5), 616–627.

Oliphant, A. J., Wynne, R. H., Zipper, C. E., Ford, W. M., Donovan, P. F., & Li, J. (2017). Autumn olive *(Elaeagnus umbellata)* presence and proliferation on former surface coal mines in Eastern USA. *Biological Invasions, 19*(1), 179–195. https://doi.org/10.1007/s10530-016-1271-6

Oliveira, S. V. de, Vargas, A., Rocha, S. M., Pereira, L. R. M., Oliveira, C. G. de, & Silva, V. S. (2018). The nature of attacks by wild boar *(Sus scrofa)* and wild boar/domestic pig hybrids *('javaporcos')* and the conduct of anti-rabies care in Brazil. *InterAmerican Journal of Medicine and Health, 1*, e-201801001. https://doi.org/10.31005/iajmh.v1i1.19

O'Loughlin, L. S., & Green, P. T. (2017). Secondary invasion: When invasion success is contingent on other invaders altering the properties of recipient ecosystems. *Ecology and Evolution, 7*(19), 7628–7637. https://doi.org/10.1002/ece3.3315

Oppenheimer, M., Glavovic, B., Hinkel, J., van de Wal, R., Magnan, A. K., Abd-Elgawad, A., Cai, R., Cifuentes-Jara, M., Deconto, R. M., & Ghosh, T. (2019). Sea level rise and implications for low lying islands, coasts and communities. In *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate* [H.-O. Pörtner, D.C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, A. Alegría, M. Nicolai, A. Okem, J. Petzold, B. Rama, N.M. Weyer (eds.)]. The Intergovernmental Panel on Climate Change. https://repositorio.catie.ac.cr/handle/11554/9280

Orsenigo, S., Mondoni, A., Rossi, G., & Abeli, T. (2014). Some like it hot and some like it cold, but not too much: Plant responses to climate extremes. *Plant Ecology, 215*(7), 677–688. https://doi.org/10.1007/s11258-014-0363-6

Ortega, J. C. G., Júlio, H. F., Gomes, L. C., & Agostinho, A. A. (2015). Fish farming as the main driver of fish introductions in Neotropical reservoirs. *Hydrobiologia, 746*(1), 147–158. https://doi.org/10.1007/s10750-014-2025-z

Ortiz-Alcaraz, A., Arnaud, G., Aguirre-Muñoz, A., Galina-Tessaro, P., Méndez-Sánchez, F., Ortega-Rubio, A., Ortiz-Alcaraz, A., Arnaud, G., Aguirre-Muñoz, A., Galina-Tessaro, P., Méndez-Sánchez, F., & Ortega-Rubio, A. (2017). Dieta y ámbito hogareño del gato feral, *Felis catus* (Carnivora: Felidae) en la Isla Socorro, Archipiélago de Revillagigedo, México. *Acta Zoológica Mexicana, 33*(3), 482–489. http://www.scielo.org.mx/scielo.php?script=sci_abstract&pid=S0065-17372017000300482&lng=es&nrm=iso&tlng=pt

Osborne, K., & Poynton, H. (2019). Copper pollution enhances the competitive advantage of invasive ascidians. *Management of Biological Invasions, 10*(4), 641–656. https://doi.org/10.3391/mbi.2019.10.4.05

Osterman, J., Aizen, M. A., Biesmeijer, J. C., Bosch, J., Howlett, B. G., Inouye, D. W., Jung, C., Martins, D. J., Medel, R., Pauw, A., Seymour, C. L., & Paxton, R. J. (2021). Global trends in the number and diversity of managed pollinator species. *Agriculture, Ecosystems & Environment, 322*, 107653. https://doi.org/10.1016/j.agee.2021.107653

Özcan, T., & Galil, B. (2006). *The first record of the banana prawn Fenneropenaeus merguiensis (De Man, 1888) (Crustacea: Decapoda: Penaeidae) from the Mediterranean Sea.* https://doi.org/10.3391/AI.2006.1.4.13

Paap, T., Wingfield, M. J., Burgess, T. I., Hulbert, J. M., & Santini, A. (2020). Harmonising the fields of invasion science and forest pathology. *NeoBiota, 62*, 301–332. https://doi.org/10.3897/neobiota.62.52991

Padayachee, A. L., Irlich, U. M., Faulkner, K. T., Gaertner, M., Procheş, Ş., Wilson, J. R. U., & Rouget, M. (2017). How do invasive species travel to and through urban environments? *Biological Invasions, 19*(12), 3557–3570. https://doi.org/10.1007/s10530-017-1596-9

Padilha, D. L., Loregian, A. C., & Budke, J. C. (2015). Forest fragmentation does not matter to invasions by Hovenia dulcis. *Biodiversity and Conservation, 24*(9), 2293–2304. https://doi.org/10.1007/s10531-015-0930-8

Padilla, D. K., & Williams, S. L. (2004). Beyond ballast water: Aquarium and ornamental trades as sources of invasive species in aquatic ecosystems. *Frontiers in Ecology and the Environment, 2*(3), 131–138. https://doi.org/10.1890/1540-9295(2004)002[0131:BBWAAO]2.0.CO;2

Pallewatta, N., Reaser, J. K., & Gutierrez, A. T. (Eds.). (2003). Sri Lanka. In *Invasive Alien Species in South-Southeast Asia: National Reports & Directory of Resources* (p. 111). Global Invasive Species Programme. https://www.doi.gov/sites/doi.gov/files/uploads/invasive_alien_species_in_south_southeast_asia.pdf

Pándi, I., Penksza, K., Botta-Dukát, Z., & Kröel-Dulay, G. (2014). People move but cultivated plants stay: Abandoned farmsteads support the persistence and spread of alien plants. *Biodiversity and Conservation, 23*(5), 1289–1302. https://doi.org/10.1007/s10531-014-0665-y

Panzavolta, T., Bracalini, M., Benigno, A., & Moricca, S. (2021). Alien Invasive Pathogens and Pests Harming Trees, Forests, and Plantations: Pathways, Global Consequences and Management. *Forests, 12*(10), Article 10. https://doi.org/10.3390/f12101364

Papavlasopoulou, I., Perdikaris, C., Vardakas, L., & Paschos, I. (2014). Enemy at the gates: Introduction potential of non-indigenous freshwater crayfish in Greece *via* the aquarium trade. *Open Life Sciences, 9*(1), 11–18. https://doi.org/10.2478/s11535-013-0120-6

Pardo, L. H., Fenn, M. E., Goodale, C. L., Geiser, L. H., Driscoll, C. T., Allen, E. B., Baron, J. S., Bobbink, R., Bowman, W. D., Clark, C. M., Emmett, B., Gilliam, F. S., Greaver, T. L., Hall, S. J., Lilleskov, E. A., Liu, L., Lynch, J. A., Nadelhoffer, K. J., Perakis, S. S., … Dennis, R. L. (2011). Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. *Ecological Applications, 21*(8), Article 8. https://doi.org/10.1890/10-2341.1

Parker, J. D., & Hay, M. E. (2005). Biotic resistance to plant invasions? Native herbivores prefer non-native plants. *Ecology Letters, 8*(9), 959–967. https://doi.org/10.1111/j.1461-0248.2005.00799.x

Parmesan, C., & Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *Nature, 421*(6918), Article 6918. https://doi.org/10.1038/nature01286

Pasiecznik, N. M., Choge, S. K., Rosenfeld, A. B., & Harris, P. J. C. (2007). Underutilised crops for famine and poverty alleviation: A case study on the potential of the multipurpose *Prosopis* tree. *5th International Symposium on New Crops and Uses: Their Role in a Rapidly Changing World*, 326–346. https://eprints.soton.ac.uk/74294/1/New.pdf#page=329

Pauchard, A., Milbau, A., Albihn, A., Alexander, J., Burgess, T., Daehler, C., Englund, G., Essl, F., Evengård, B., Greenwood, G. B., Haider, S., Lenoir, J., McDougall, K., Muths, E., Nuñez, M. A., Olofsson, J., Pellissier, L., Rabitsch, W., Rew, L. J., … Kueffer, C. (2016). Non-native and native organisms moving into high elevation and high latitude ecosystems in an era of climate change: New challenges for ecology and conservation. *Biological Invasions, 18*(2), 345–353. https://doi.org/10.1007/s10530-015-1025-x

Pausas, J. G., & Keeley, J. E. (2019). Wildfires as an ecosystem service. *Frontiers in Ecology and the Environment, 17*(5), Article 5. https://doi.org/10.1002/fee.2044

Pautasso, M., Aas, G., Queloz, V., & Holdenrieder, O. (2013). European ash *(Fraxinus excelsior)* dieback–A conservation biology challenge. *Biological Conservation, 158*, 37–49.

Pendrill, F., Persson, U. M., Godar, J., Kastner, T., Moran, D., Schmidt, S., & Wood, R. (2019). Agricultural and forestry trade drives large share of tropical deforestation emissions. *Global Environmental Change-Human and Policy Dimensions, 56*, 1–10. https://doi.org/10.1016/j.gloenvcha.2019.03.002

Pereira, H. M., Leadley, P. W., Proença, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarrés, J. F., Araújo, M. B., Balvanera, P., Biggs, R., Cheung, W. W. L., Chini, L., Cooper, H. D., Gilman, E. L., Guénette, S., Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., … Walpole, M. (2010). Scenarios for Global Biodiversity in the 21st Century. *Science, 330*(6010), 1496–1501. https://doi.org/10.1126/science.1196624

Peres, C. K., Lambrecht, R. W., Tavares, D. A., & Chiba de Castro, W. A. (2018). Alien Express: The threat of aquarium e-commerce introducing invasive aquatic plants in Brazil. *Perspectives in Ecology and Conservation, 16*(4), 221–227. https://doi.org/10.1016/j.pecon.2018.10.001

Pérez, J. E., Alfonsi, C., Nirchio, M., Muñoz, C., & Gómez, J. A. (2003). The introduction of exotic species in aquaculture: A solution or part of the problem? *Interciencia, 28*(4), 234–238. https://www.researchgate.net/publication/233808881_The_introduction_of_exotic_species_in_aquaculture_A_solution_or_part_of_the_problem

Perrings, C. (2010). Exotic effects of capital accumulation. *Proceedings of the National Academy of Sciences, 107*(27), Article 27. https://doi.org/10.1073/pnas.1007335107

Perrings, C., Dehnen-Schmutz, K., Touza, J., & Williamson, M. (2005). How to manage biological invasions under globalization. *Trends in Ecology & Evolution, 20*(5), 212–215. https://doi.org/10.1016/j.tree.2005.02.011

Persley, D. M., & Syme, J. R. (1990). *Field Crops and Pastures: A Disease Management Guide.* Queensland Department of Primary Industries, Brisbane, Australia.

Pesante, D. G., Rinderer, T. E., & Collins, A. M. (1987). Differential Pollen Collection by Africanized and European Honeybees in Venezuela. *Journal of Apicultural Research, 26*(1), 24–29. https://doi.org/10.1080/00218839.1987.11100730

Pessel, D., Lecomte, J., Emeriau, V., Krouti, M., Messean, A., & Gouyon, P. H. (2001). Persistence of oilseed rape (*Brassica napus* L.) outside of cultivated fields. *Theoretical and Applied Genetics, 102*(6), 841–846. https://doi.org/10.1007/s001220100583

Petrick, S., Riemann-Campe, K., Hoog, S., Growitsch, C., Schwind, H., Gerdes, R., & Rehdanz, K. (2017). Climate change, future Arctic Sea ice, and the competitiveness of European Arctic offshore oil and gas production on world markets. *Ambio, 46*(3), 410–422. https://doi.org/10.1007/s13280-017-0957-z

Petrides, G. A. (1975). The importation of wild ungulates into Latin America, with remarks on their environmental effects. *Environmental Conservation, 2*(1), 47–51. https://doi.org/10.1017/S0376892900000667

Pfeifer-Meister, L., Cole, E. M., Roy, B. A., & Bridgham, S. D. (2008). Abiotic constraints on the competitive ability of exotic and native grasses in a Pacific Northwest prairie. *Oecologia, 155*(2), 357–366. https://doi.org/10.1007/s00442-007-0909-6

Phillips, M. L., Murray, B. R., Pyšek, P., Pergl, J., Jaroš ík, V., Chytrý, M., & Kühn, I. (2010). *Plant species of the Central European flora as aliens in Australia.*

Pickering, C., & Hill, W. (2007). Roadside weeds of the snowy mountains, Australia. *Mountain Research and Development, 27*(4), 359–367. https://doi.org/10.1659/mrd.0805

Pickering, C., & Mount, A. (2010). Do tourists disperse weed seed? A global review of unintentional human-mediated terrestrial seed dispersal on clothing, vehicles and horses. *Journal of Sustainable Tourism, 18*(2), 239–256. https://doi.org/10.1080/09669580903406613

Pienimäki, M., & Leppäkoski, E. (2004). Invasion Pressure on the Finnish Lake District: Invasion Corridors and Barriers. *Biological Invasions, 6*(3), 331–346. https://doi.org/10.1023/B:BINV.0000034607.00490.95

Pietsch, M. (2005). *Soil and biowaste as potential pathways for invasive alien species, and regulatory approaches to minimize the risks. 81.*

Pilcher, H. R. (2004). US army may have killed Italian trees. *Nature*, news040405-1. https://doi.org/10.1038/news040405-1

Pimentel, D., Acquay, H., Biltonen, M., Rice, P., Silva, M., Nelson, J., Lipner, V., Giordano, S., Horowitz, A., & D'Amore, M. (1992). Environmental and Economic Costs of Pesticide Use. *BioScience, 42*(10), 750–760. https://doi.org/10.2307/1311994

Pimentel, D., McNair, S., Janecka, J., Wightman, J., Simmonds, C., O'Connell, C., Wong, E., Russel, L., Zern, J., Aquino, T., & Tsomondo, T. (2001). Economic and environmental threats of alien plant, animal, and microbe invasions. *Agriculture, Ecosystems & Environment, 84*(1), Article 1. https://doi.org/10.1016/S0167-8809(00)00178-X

Pinochet, J., Urbina, M. A., & Lagos, M. E. (2020). Marine invertebrate larvae love plastics: Habitat selection and settlement on artificial substrates. *Environmental Pollution, 257*, 113571. https://doi.org/10.1016/j.envpol.2019.113571

Piola, R. F., & Johnston, E. L. (2008). Pollution reduces native diversity and increases invader dominance in marine hard-substrate communities. *Diversity and Distributions, 14*(2), 329–342. https://doi.org/10.1111/j.1472-4642.2007.00430.x

Piria, M., Simonović, P., Kalogianni, E., Vardakas, L., Koutsikos, N., Zanella, D., Ristovska, M., Apostolou, A., Adrović, A., Mrdak, D., Tarkan, A. S., Milošević, D., Zanella, L. N., Bakiu, R., Ekmekçi, F. G., Povž, M., Korro, K., Nikolić, V., Škrijelj, R., … Joy, M. K. (2018). Alien freshwater fish species in the Balkans-Vectors and pathways of introduction. *Fish and Fisheries, 19*(1), 138–169. https://doi.org/10.1111/faf.12242

Pirotta, V., Grech, A., Jonsen, I. D., Laurance, W. F., & Harcourt, R. G. (2019). Consequences of global shipping traffic for marine giants. *Frontiers in Ecology and the Environment, 17*(1), 39–47. https://doi.org/10.1002/fee.1987

Plaza, P. I., & Lambertucci, S. A. (2017). How are garbage dumps impacting vertebrate demography, health, and conservation? *Global Ecology and Conservation, 12*, 9–20. https://doi.org/10.1016/j.gecco.2017.08.002

Plaza, P. I., Speziale, K. L., & Lambertucci, S. A. (2018). Rubbish dumps as invasive plant epicentres. *Biological Invasions, 20*(9), 2277–2283. https://doi.org/10.1007/s10530-018-1708-1

Policelli, N., Bruns, T. D., Vilgalys, R., & Nuñez, M. A. (2019). Suilloid fungi as global drivers of pine invasions. *New Phytologist, 222*(2), 714–725. https://doi.org/10.1111/nph.15660

Policelli, N., Vietorisz, C., Bhatnagar, J. M., & Nuñez, M. A. (2022). Ectomycorrhizal Fungi Invasions in Southern South America. In M. A. Lugo & M. C. Pagano (Eds.), *Mycorrhizal Fungi in South America* (pp. 25–46). Springer International Publishing. https://doi.org/10.1007/978-3-031-12994-0_2

Poorter, H., & Navas, M. (2003). Plant growth and competition at elevated CO 2: On winners, losers and functional groups. *New Phytologist, 157*(2), 175–198. https://doi.org/10.1046/j.1469-8137.2003.00680.x

Pörtner, H.-O., Scholes, R. J., Agard, J., Archer, E., Bai, X., Barnes, D., Burrows, M., Chan, L., Cheung, W. L. (William), Diamond, S., Donatti, C., Duarte, C., Eisenhauer, N., Foden, W., Gasalla, M. A., Handa, C., Hickler, T., Hoegh-Guldberg, O., Ichii, K., … Ngo, H. (2021). *IPBES-IPCC co-sponsored workshop report on biodiversity and climate change* (Version 2). Zenodo. https://doi.org/10.5281/ZENODO.4782538

Prabakaran, K., Li, J., Anandkumar, A., Leng, Z., Zou, C. B., & Du, D. (2019). Managing environmental contamination through phytoremediation by invasive plants: A review. *Ecological Engineering, 138*, 28–37. https://doi.org/10.1016/j.ecoleng.2019.07.002

Pratt, C., & Lottermoser, B. G. (2007). Trace metal uptake by the grass *Melinis repens* from roadside soils and sediments, tropical Australia. *Environmental Geology, 52*(8), 1651–1662. https://doi.org/10.1007/s00254-006-0610-9

Pratt, T. C., O'Connor, L. M., Stacey, J. A., Stanley, D. R., Mathers, A., Johnson, L. E., Reid, S. M., Verreault, G., & Pearce, J. (2019). Pattern of *Anguillicoloides crassus* infestation in the St. Lawrence River watershed. *Journal of Great Lakes Research, 45*(5), 991–997. https://doi.org/10.1016/j.jglr.2019.06.005

Puckett, E. E., & Munshi-South, J. (2019). Brown rat demography reveals pre-commensal structure in eastern Asia prior to expansion into Southeast Asia. *Genome Research*, gr.235754.118. https://doi.org/10.1101/gr.235754.118

Puechmaille, S. J., Frick, W. F., Kunz, T. H., Racey, P. A., Voigt, C. C., Wibbelt, G., & Teeling, E. C. (2011). White-nose syndrome: Is this emerging disease a threat to European bats? *Trends in Ecology & Evolution, 26*(11), 570–576. https://doi.org/10.1016/j.tree.2011.06.013

Puth, L. M., & Post, D. M. (2005). Studying invasion: Have we missed the boat? *Ecology Letters, 8*(7), 715–721. https://doi.org/10.1111/j.1461-0248.2005.00774.x

Pyšek, P., Bacher, S., Chytrý, M., Jarošík, V., Wild, J., Celesti-Grapow, L., Gassó, N., Kenis, M., Lambdon, P. W., Nentwig, W., Pergl, J., Roques, A., Sádlo, J., Solarz, W., Vilà, M., & Hulme, P. E. (2010). Contrasting patterns in the invasions of European terrestrial and freshwater habitats by alien plants, insects and vertebrates. *Global Ecology and Biogeography, 19*(3), 317–331. https://doi.org/10.1111/j.1466-8238.2009.00514.x

Pyšek, P., Hulme, P. E., Simberloff, D., Bacher, S., Blackburn, T. M., Carlton, J. T., Dawson, W., Essl, F., Foxcroft, L. C., Genovesi, P., Jeschke, J. M., Kühn, I., Liebhold, A. M., Mandrak, N. E., Meyerson, L. A., Pauchard, A., Pergl, J., Roy, H. E., Seebens, H., … Richardson, D. M. (2020). Scientists' warning on invasive alien species. *Biological Reviews, 95*(6), 1511–1534. https://doi.org/10.1111/brv.12627

Pyšek, P., Jarosik, V., Hulme, P. E., Kühn, I., Wild, J., Arianoutsou, M., Bacher, S., Chiron, F., Didžiulis, V., Essl, F., Genovesi, P., Gherardi, F., Hejda, M., Kark, S., Lambdon, P. W., Desprez-Loustau, M.-L., Nentwig, W., Pergl, J., Poboljšaj, K., … Winter, M. (2010). Disentangling the role of environmental and human pressures on biological invasions across Europe. *Proceedings of the National Academy of Sciences, 107*(27), 12157–12162. https://doi.org/10.1073/pnas.1002314107

Pyšek, P., Jarošík, V., Müllerová, J., Pergl, J., & Wild, J. (2008). Comparing the rate of invasion by *Heracleum mantegazzianum* at continental, regional, and local scales. *Diversity and Distributions, 14*(2), 355–363. https://doi.org/10.1111/j.1472-4642.2007.00431.x

Pyšek, P., & Prach, K. (1996). Plant Invasions and the Role of Riparian Habitats: A Comparison of Four Species Alien to Central Europe. In F. B. Samson & F. L. Knopf (Eds.), *Ecosystem Management: Selected Readings* (pp. 254–263). Springer. https://doi.org/10.1007/978-1-4612-4018-1_23

Quinn, L. D., Gordon, D. R., Glaser, A., Lieurance, D., & Flory, S. L. (2015). Bioenergy Feedstocks at Low Risk for Invasion in the USA: A "White List" Approach. *BioEnergy Research, 8*(2), 471–481. https://doi.org/10.1007/s12155-014-9503-z

Raffaele, E., Nuñez, M. A., Eneström, J., & Blackhall, M. (2016). Fire as mediator of pine invasion: Evidence from Patagonia, Argentina. *Biological Invasions, 18*(3), 597–601. https://doi.org/10.1007/s10530-015-1038-5

Rahel, F. J. (2002). Homogenization of Freshwater Faunas. *Annual Review of Ecology and Systematics, 33*(1), 291–315. https://doi.org/10.1146/annurev.ecolsys.33.010802.150429

Rahlao, S. J., Milton, S. J., Esler, K. J., & Barnard, P. (2014). Performance of invasive alien fountain grass (*Pennisetum setaceum)* along a climatic gradient through three South African biomes. *South African Journal of Botany, 91*, 43–48. https://doi.org/10.1016/j.sajb.2013.11.013

Rahlao, S. J., Milton, S. J., Esler, K. J., Wilgen, B. W., & Barnard, P. (2009). Effects of invasion of fire-free arid shrublands by a fire-promoting invasive alien grass *(Pennisetum setaceum)* in South Africa. *Austral Ecology, 34*(8), 920–928. https://doi.org/10.1111/j.1442-9993.2009.02000.x

Rahman, M. L., Tarrant, S., Ollerton, J., & McCollin, D. (2016). Effect of soil conditions and landscape factors on macro-snail communities in newly created grasslands of restored landfill sites in the UK. *Zoology and Ecology, 26*(4), 275–281. https://doi.org/10.1080/21658005.2016.1216102

Rakauskas, V., Stakėnas, S., Virbickas, T., & Bukelskis, E. (2016). Non-indigenous fish in the northern branch of the central European invasion corridor. *Reviews in Fish Biology and Fisheries, 26*(3), 491–508. https://doi.org/10.1007/s11160-016-9438-x

Rakha, M., Zekeya, N., Sevgan, S., Musembi, M., Ramasamy, S., & Hanson, P. (2017). Screening recently identified whitefly/spider mite-resistant wild tomato accessions for resistance to *Tuta absoluta. Plant Breeding, 136*(4), 562–568. https://doi.org/10.1111/pbr.12503

Rand, T. A., & Tscharntke, T. (2007). Contrasting effects of natural habitat loss on generalist and specialist aphid natural enemies. *Oikos, 116*(8), 1353–1362. https://doi.org/10.1111/j.0030-1299.2007.15871.x

Randall, R. P. (2017). *A global compendium of weeds.* (Ed.3). R. P. Randall. https://www.cabdirect.org/cabdirect/abstract/20173071957

Randerson, J. (2003, June 27). *Fungus in your tea, sir?* New Scientist. https://www.newscientist.com/article/mg17824010-900-fungus-in-your-tea-sir/

Rashid, I., Haq, S. M., Lembrechts, J. J., Khuroo, A. A., Pauchard, A., & Dukes, J. S. (2021). Railways redistribute plant species in mountain landscapes. *Journal of Applied Ecology, 58*(9), 1967–1980. https://doi.org/10.1111/1365-2664.13961

Ravi, S., D'Odorico, P., Collins, S. L., & Huxman, T. E. (2009). Can Biological Invasions Induce Desertification? *The New Phytologist, 181*(3), 512–515. https://www.jstor.org/stable/30224699

Reaser, J. K. (2003). Invasive alien species: Definitions, causes, consequences, and the way forward. In N. Pallewatta, J. K. Reaser, & A. Gutierrez (Eds.), *Prevention and Management of Invasive Alien Species: Proceedings of a Workshop on Forging Cooperation throughout South and Southeast Asia* (pp. 11–18). Global Invasive Species Programme. https://giasipartnership.myspecies.info/en/content/pallewatta-n-jk-reaser-gutierrez-eds-2003-prevention-and-management-invasive-alien-species

Rebbeck, J., Hutchinson, T., Iverson, L., Yaussy, D., & Fox, T. (2017). Distribution and demographics of *Ailanthus altissima* in an oak forest landscape managed with timber harvesting and prescribed fire. *Forest Ecology and Management, 401*, 233–241. https://doi.org/10.1016/j.foreco.2017.06.050

Rech, S., Borrell, Y., & Garcia-Vazquez, E. (2016). Marine litter as a vector for non-native species: What we need to know. *Marine Pollution Bulletin, 113*(1–2), 40–43. https://doi.org/10.1016/j.marpolbul.2016.08.032

Rech, S., Pichs, Y., & Garcia-Vasquez, E. (2018). Anthropogenic marine litter composition in coastal areas may be a predictor of potentially invasive rafting fauna. *PLoS ONE, 13*(1), 1–22. https://doi.org/10.1371/journal.pone.0191859

Reichman, J. R., Watrud, L. S., Lee, E. H., Burdick, C. A., Bollman, M. A., Storm, M. J., King, G. A., & Mallory-Smith, C. (2006). Establishment of transgenic herbicide-resistant creeping bentgrass (*Agrostis stolonifera* L.) in nonagronomic habitats. *Molecular Ecology, 15*(13), 4243–4255. https://doi.org/10.1111/j.1365-294X.2006.03072.x

Reise, K., Gollasch, S., & Wolff, W. J. (1998). Introduced marine species of the North Sea coasts. *Helgolander Meeresunters, 52*(3–4), 219–234. https://doi.org/10.1007/BF02908898

Renault, D., Laparie, M., McCauley, S. J., & Bonte, D. (2018). Environmental Adaptations, Ecological Filtering, and Dispersal Central to Insect Invasions. *Annual Review of Entomology, 63*(1), 345–368. https://doi.org/10.1146/annurev-ento-020117-043315

Rentch, J. S., Fortney, R. H., Stephenson, S. L., Adams, H. S., Grafton, W. N., & Anderson, J. T. (2005). Vegetation–site relationships of roadside plant communities in West Virginia, USA. *Journal of Applied Ecology, 42*(1), 129–138. https://doi.org/10.1111/j.1365-2664.2004.00993.x

Rew, L. J., Brummer, T. J., Pollnac, F. W., Larson, C. D., Taylor, K. T., Taper, M. L., Fleming, J. D., & Balbach, H. E. (2018). Hitching a ride: Seed accrual rates on different types of vehicles. *Journal of Environmental Management, 206*, 547–555. https://doi.org/10.1016/j.jenvman.2017.10.060

Reynolds, C., Miranda, N. A. F., & Cumming, G. S. (2015). The role of waterbirds in the dispersal of aquatic alien and invasive species. *Diversity and Distributions, 21*(7), 744–754. https://doi.org/10.1111/ddi.12334

Rhodes, A. C., Plowes, R. M., Goolsby, J. A., Gaskin, J. F., Musyoka, B., Calatayud, P.-A., Cristofaro, M., Grahmann, E. D., Martins, D. J., & Gilbert, L. E. (2021). The dilemma of Guinea grass *(Megathyrsus maximus)*: A valued pasture grass and a highly invasive species. *Biological Invasions, 23*(12), Article 12. https://doi.org/10.1007/s10530-021-02607-3

Rhymer, J. M., Williams, M. J., & Braun, M. J. (1994). Mitochondrial Analysis of Gene Flow between New Zealand Mallards *(Anas Platyrhynchos)* and Grey Ducks *(A. Superciliosa). The Auk, 111*(4), Article 4. https://doi.org/10.2307/4088829

Ribeiro, F., Elvira, B., Collares-Pereira, M. J., & Moyle, P. B. (2008). Life-history traits of non-native fishes in Iberian watersheds across several invasion stages: A first approach. *Biological Invasions, 10*(1), 89–102. https://doi.org/10.1007/s10530-007-9112-2

Ribeiro, V. R., Silva, P. R. L. da, Gubiani, É. A., Faria, L., Daga, V. S., & Vitule, J. R. S. (2017). Imminent threat of the predator fish invasion *Salminus brasiliensis* in a Neotropical ecoregion: Eco-vandalism masked as an environmental project. *Perspectives in Ecology and Conservation, 15*(2), 132–135. https://doi.org/10.1016/j.pecon.2017.03.004

Ricciardi, A. (2001). Facilitative interactions among aquatic invaders: Is an “invasional meltdown” occurring in the Great Lakes? *Canadian Journal of Fisheries and Aquatic Sciences, 58*(12), 2513–2525. https://doi.org/10.1139/f01-178

Ricciardi, A. (2007). Are Modern Biological Invasions an Unprecedented Form of Global Change? *Conservation Biology, 21*(2), 329–336. https://doi.org/10.1111/j.1523-1739.2006.00615.x

Ricciardi, A., Blackburn, T. M., Carlton, J. T., Dick, J. T. A., Hulme, P. E., Iacarella, J. C., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Sutherland, W. J., Wardle, D. A., & Aldridge, D. C. (2017). Invasion Science: A Horizon Scan of Emerging Challenges and Opportunities. *Trends in Ecology & Evolution, 32*(6), 464–474. https://doi.org/10.1016/j.tree.2017.03.007

Ricciardi, A., Iacarella, J. C., Aldridge, D. C., Blackburn, T. M., Carlton, J. T., Catford, J. A., Dick, J. T. A., Hulme, P. E., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Meyerson, L. A., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Vilà, M., & Wardle, D. A. (2020). Four priority areas to advance invasion science in the face of rapid environmental change. *Environmental Reviews, 29*(2), 119–141. https://doi.org/10.1139/er-2020-0088

Ricciardi, A., & Simberloff, D. (2009). Assisted colonization is not a viable conservation strategy. *Trends in Ecology & Evolution, 24*(5), 248–253. https://doi.org/10.1016/j.tree.2008.12.006

Richardson, D. M. (1998). Forestry Trees as Invasive Aliens. *Conservation Biology, 12*(1), 18–26. https://doi.org/10.1046/j.1523-1739.1998.96392.x

Richardson, D. M. (2006). *Pinus*: A model group for unlocking the secrets of alien plant invasions? *Preslia, 78*, 375–388. https://www.preslia.cz/P064CRic.pdf

Richardson, D. M., Allsopp, N., D’Antonio, C. M., Milton, S. J., & Rejmánek, M. (2007). Plant invasions—The role of mutualisms. *Biological Reviews, 75*(1), 65–93. https://doi.org/10.1111/j.1469-185X.1999.tb00041.x

Richardson, D. M., & Blanchard, R. (2011). Learning from our mistakes: Minimizing problems with invasive biofuel plants. *Current Opinion in Environmental Sustainability, 3*(1–2), Article 1–2. https://doi.org/10.1016/j.cosust.2010.11.006

Richardson, D. M., Carruthers, J., Hui, C., Impson, F. A. C., Miller, J. T., Robertson, M. P., Rouget, M., Le Roux, J. J., & Wilson, J. R. U. (2011). Human-mediated introductions of Australian *acacias* – a global experiment in biogeography. *Diversity and Distributions, 17*(5), 771–787. https://doi.org/10.1111/j.1472-4642.2011.00824.x

Richardson, D. M., Hellmann, J. J., McLachlan, J. S., Sax, D. F., Schwartz, M. W., Gonzalez, P., Brennan, E. J., Camacho, A., Root, T. L., Sala, O. E., Schneider, S. H., Ashe, D. M., Clark, J. R., Early, R., Etterson, J. R., Fielder, E. D., Gill, J. L., Minteer, B. A., Polasky, S., … Vellend, M. (2009). Multidimensional evaluation of managed relocation. *Proceedings of the National Academy of Sciences, 106*(24), 9721–9724. https://doi.org/10.1073/pnas.0902327106

Richardson, D. M., Holmes, P. M., Esler, K. J., Galatowitsch, S. M., Stromberg, J. C., Kirkman, S. P., Pyšek, P., & Hobbs, R. J. (2007). Riparian vegetation: Degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions, 13*(1), 126–139. https://doi.org/10.1111/j.1366-9516.2006.00314.x

Richardson, D. M., Le Roux, J. J., & Wilson, J. R. (2015). Australian *acacias* as invasive species: Lessons to be learnt from regions with long planting histories. *Southern Forests: A Journal of Forest Science, 77*(1), Article 1. https://doi.org/10.2989/20702620.2014.999305

Richardson, D. M., & Nsikani, M. M. (2021). Mediterranean Pines as Invasive Species in the Southern Hemisphere. In G. Ne’eman & Y. Osem (Eds.), *Pines and Their Mixed Forest Ecosystems in the Mediterranean Basin* (pp. 83–99). Springer International Publishing. https://doi.org/10.1007/978-3-030-63625-8_5

Richardson, D. M., & Rejmánek, M. (2011). Trees and shrubs as invasive alien species – a global review: Global review of invasive trees & shrubs. *Diversity and Distributions, 17*(5), Article 5. https://doi.org/10.1111/j.1472-4642.2011.00782.x

Richardson, D. M., Witt, A. B. R., Pergl, J., Dawson, W., Essl, F., Kreft, H., van Kleunen, M., Weigelt, P., Winter, M., & Pyšek, P. (2021). Plant invasions in Africa. In D. R. Clements, M. K. Upadhyaya, S. Joshi, & A. Shrestha (Eds.), *Global Plant Invasions* (pp. 225–252). Springer. https://doi.org/10.1007/978-3-030-89684-3_11

Rigaud, K. K., de Sherbinin, A., Jones, B., Bergmann, J., Clement, V., Ober, K., Schewe, J., Adamo, S., McCusker, B., Heuser, S., & Midgley, A. (2018). *Groundswell: Preparing for Internal Climate Migration. World Bank*. http://hdl.handle.net/10986/29461

Rigling, D., & Prospero, S. (2018). *Cryphonectria parasitica*, the causal agent of chestnut blight: Invasion history, population biology and disease control. *Molecular Plant Pathology, 19*(1), 7–20.

Rignot, E., Velicogna, I., van den Broeke, M. R., Monaghan, A., & Lenaerts, J. T. M. (2018). Acceleration of the contribution of the Greenland and Antarctic ice sheets to sea level rise. *Geophysical Research Letters*. https://doi.org/10.1029/2011GL046583@10.1002/(ISSN)1944-8007.GRL40

Riley, S. (2005). Invasive Alien Species and the Protection of Biodiversity: The Role of Quarantine Laws in Resolving Inadequacies in the International Legal Regime. *Journal of Environmental Law, 17*(3), 323–359. https://doi.org/10.1093/jel/eqi028

Riley, S. J., & Banks, R. G. (1996). The role of phosphorus and heavy metals in the spread of weeds in urban bushlands: An example from the Lane Cove Valley, NSW, Australia. *Science of The Total Environment, 182*(1–3), 39–52. https://doi.org/10.1016/0048-9697(95)05033-7

Rilov, G., Peleg, O., Yeruham, E., Garval, T., Vichik, A., & Raveh, O. (2018). Alien turf: Overfishing, overgrazing and invader domination in south-eastern Levant reef ecosystems. *Aquatic Conservation: Marine and Freshwater Ecosystems, 28*(2), 351–369. https://doi.org/10.1002/aqc.2862

Río-Rama, M., Maldonado-Erazo, C., Durán-Sánchez, A., & Álvarez-García, J. (2019). Mountain tourism research. A review. *European Journal of Tourism Research, 22*(2019), 130–150. https://www.semanticscholar.org/paper/Mountain-tourism-research.-A-review-R%C3%ADo-Rama-Maldonado-Erazo/14a7b727a2bb05d19fd7f2a8f4ac97f5e36b79bb

Rizzo, D. M., & Garbelotto, M. (2003). Sudden oak death: Endangering California and Oregon forest ecosystems. *Frontiers in Ecology and the Environment, 1*(4), 197–204. https://doi.org/10.1890/1540-9295(2003)001[0197:SODECA]2.0.CO;2

Roberge, C., Normandeau, É., Einum, S., Guderley, H., & Bernatchez, L. (2008). Genetic consequences of interbreeding between farmed and wild Atlantic salmon: Insights from the transcriptome. *Molecular Ecology, 17*(1), 314–324. https://doi.org/10.1111/j.1365-294X.2007.03438.x

Roberts, J., & Florentine, S. (2021). Biology, distribution and control of the invasive species *Ulex europaeus* (Gorse): A global synthesis of current and future management challenges and research gaps. *Weed Research, 61*(4), Article 4. https://doi.org/10.1111/wre.12491

Rödder, D., Schmidtlein, S., Veith, M., & Lötters, S. (2009). Alien Invasive Slider Turtle in Unpredicted Habitat: A Matter of Niche Shift or of Predictors Studied? *PLoS ONE, 4*(11), e7843. https://doi.org/10.1371/journal.pone.0007843

Rodewald, A. D., & Arcese, P. (2016). Direct and Indirect Interactions between Landscape Structure and Invasive or Overabundant Species. *Current Landscape Ecology Reports, 1*(1), 30–39. https://doi.org/10.1007/s40823-016-0004-y

Roiz, D., Neteler, M., Castellani, C., Arnoldi, D., & Rizzoli, A. (2011). Climatic Factors Driving Invasion of the Tiger Mosquito (*Aedes albopictus*) into New Areas of Trentino, Northern Italy. *PLoS ONE, 6*(4), e14800. https://doi.org/10.1371/journal.pone.0014800

Rojas-Sandoval, J., & Acevedo-Rodríguez, P. (2015). Naturalization and invasion of alien plants in Puerto Rico and the Virgin Islands. *Biological Invasions, 17*(1), 149–163. https://doi.org/10.1007/s10530-014-0712-3

Rojas-Sandoval, J., Tremblay, R. L., Acevedo-Rodríguez, P., & Díaz-Soltero, H. (2017). Invasive plant species in the West Indies: Geographical, ecological, and floristic insights. *Ecology and Evolution, 7*(13), 4522–4533. https://doi.org/10.1002/ece3.2984

Rolls, R. J., Hayden, B., & Kahilainen, K. K. (2017). Conceptualising the interactive effects of climate change and biological invasions on subarctic freshwater fish. *Ecology and Evolution, 7*(12), 4109–4128. https://doi.org/10.1002/ece3.2982

Romañach, S. S., Beerens, J. M., Patton, B. A., Chapman, J. P., & Hanson, M. R. (2019). Impacts of Saltwater Intrusion on Wetland Prey Production and Composition in a Historically Freshwater Marsh. *Estuaries and Coasts, 42*(6), 1600–1611. https://doi.org/10.1007/s12237-019-00572-8

Roques, A., Auger-Rozenberg, M.-A., Blackburn, T. M., Garnas, J., Pyšek, P., Rabitsch, W., Richardson, D. M., Wingfield, M. J., Liebhold, A. M., & Duncan, R. P. (2016). Temporal and interspecific variation in rates of spread for insect species invading Europe during the last 200 years. *Biological Invasions, 18*(4), 907–920. https://doi.org/10.1007/s10530-016-1080-y

Ross, C. A., Auge, H., & Durka, W. (2008). Genetic relationships among three native North-American *Mahonia* species, invasive *Mahonia* populations from Europe, and commercial cultivars. *Plant Systematics and Evolution, 275*(3–4), 219–229. https://doi.org/10.1007/s00606-008-0066-3

Ross, L. C., Lambdon, P. W., & Hulme, P. E. (2008). Disentangling the roles of climate, propagule pressure and land use on the current and potential elevational distribution of the invasive weed *Oxalis pes-caprae*, L. on Crete. *Perspectives in Plant Ecology, Evolution and Systematics, 10*(4), 251–258. https://doi.org/10.1016/j.ppees.2008.06.001

Rouget, M., Robertson, M. P., Wilson, J. R. U., Hui, C., Essl, F., Renteria, J. L., & Richardson, D. M. (2016). Invasion debt—Quantifying future biological invasions. *Diversity and Distributions, 22*(4), 445–456. https://doi.org/10.1111/ddi.12408

Roundy, B. A., Young, K., Cline, N., Hulet, A., Miller, R. F., Tausch, R. J., Chambers, J. C., & Rau, B. (2014). Piñon–Juniper Reduction Increases Soil Water Availability of the Resource Growth Pool. *Rangeland Ecology & Management, 67*(5), 495–505. https://doi.org/10.2111/REM-D-13-00022.1

Roura-Pascual, N., Hui, C., Ikeda, T., Leday, G., Richardson, D. M., Carpintero, S., Espadaler, X., Gomez, C., Guenard, B., Hartley, S., Krushelnycky, P., Lester, P. J., McGeoch, M. A., Menke, S. B., Pedersen, J. S., Pitt, J. P. W., Reyes, J., Sanders, N. J., Suarez, A. V., … Worner, S. P. (2011). Relative roles of climatic suitability and anthropogenic influence in determining the pattern of spread in a global invader. *Proceedings of the National Academy of Sciences of the United States of America, 108*(1), 220–225. https://doi.org/10.1073/pnas.1011723108

Roura-Pascual, N., Leung, B., Rabitsch, W., Rutting, L., Vervoort, J., Bacher, S., Dullinger, S., Erb, K.-H., Jeschke, J. M., Katsanevakis, S., Kühn, I., Lenzner, B., Liebhold, A. M., Obersteiner, M., Pauchard, A., Peterson, G. D., Roy, H. E., Seebens, H., Winter, M., … Essl, F. (2021). Alternative futures for global biological invasions. *Sustainability Science, 16*(5), 1637–1650. https://doi.org/10.1007/s11625-021-00963-6

Roy, H. E., Adriaens, T., Isaac, N. J. B., Kenis, M., Onkelinx, T., Martin, G. S., Brown, P. M. J., Hautier, L., Poland, R., Roy, D. B., Comont, R., Eschen, R., Frost, R., Zindel, R., Van Vlaenderen, J., Nedvěd, O., Ravn, H. P., Grégoire, J.-C., de Biseau, J.-C., & Maes, D. (2012). Invasive alien predator causes rapid declines of native European ladybirds. *Diversity and Distributions, 18*(7), 717–725. https://doi.org/10.1111/j.1472-4642.2012.00883.x

RSPB. (2021). *The RSPB Wildlife Charity: Nature Reserves & Wildlife Conservation.* The RSPB. https://rspb.org.uk/

Rusterholz, H.-P., Wirz, D., & Baur, B. (2012). Garden waste deposits as a source for non-native plants in mixed deciduous forests. *Applied Vegetation Science, 15*(3), 329–337. https://doi.org/10.1111/j.1654-109X.2011.01175.x

Saba, A. O., Ismail, A., Zulkifli, S. Z., Shohaimi, S., Jamil, N. R., Nawi, N. M., Ghani, I. F. A., Halim, M. R. A., & Amal, M. N. A. (2020). Checklists, production trends, and potential ecological and socioeconomic impacts of non-native freshwater fishes in Malaysia: A review. *Aquatic Invasions, 15*(4), 646–670. https://doi.org/10.3391/ai.2020.15.4.07

Salomidi, M., Katsanevakis, S., Issaris, Y., Tsiamis, K., & Katsiaras, N. (2013). Anthropogenic disturbance of coastal habitats promotes the spread of the introduced scleractinian coral *Oculina patagonica* in the Mediterranean Sea. *Biological Invasions, 15*(9), 1961–1971. https://doi.org/10.1007/s10530-013-0424-0

Sánchez-Ortiz, K., Taylor, K. J. M., Palma, A. D., Essl, F., Dawson, W., Kreft, H., Pergl, J., Pyšek, P., Kleunen, M. van, Weigelt, P., & Purvis, A. (2020). Effects of land-use change and related pressures on alien and native subsets of island communities. *PLoS ONE, 15*(12), Article 12. https://doi.org/10.1371/journal.pone.0227169

Sandvik, H., Dolmen, D., Elven, R., Falkenhaug, T., Forsgren, E., Hansen, H., Hassel, K., Husa, V., Kjærstad, G., Ødegaard, F., Pedersen, H. C., Solheim, H., Stokke, B. G., Åsen, P. A., Åström, S., Brandrud, T.-E., Elven, H., Endrestøl, A., Finstad, A., … Gederaas, L. (2019). Alien plants, animals, fungi and algae in Norway: An inventory of neobiota. *Biological Invasions, 21*(10), 2997–3012. https://doi.org/10.1007/s10530-019-02058-x

Sanford, E., Gaylord, B., Hettinger, A., Lenz, E. A., Meyer, K., & Hill, T. M. (2014). Ocean acidification increases the vulnerability of native oysters to predation by invasive snails. *Proceedings of the Royal Society B-Biological Sciences, 281*(1778), 20132681. https://doi.org/10.1098/rspb.2013.2681

Santana, O. A., & Encinas, J. I. (2008). Levantamento das espécies exóticas arbóreas e seu impacto nas espécies nativas em áreas adjacentes a depósitos de resíduos domiciliares. *Biotemas, 21*(4), Article 4. https://doi.org/10.5007/2175-7925.2008v21n4p29

Santini, A., Ghelardini, L., De Pace, C., Desprez-Loustau, M. L., Capretti, P., Chandelier, A., Cech, T., Chira, D., Diamandis, S., Gaitniekis, T., Hantula, J., Holdenrieder, O., Jankovsky, L., Jung, T., Jurc, D., Kirisits, T., Kunca, A., Lygis, V., Malecka, M., … Stenlid, J. (2013). Biogeographical patterns and determinants of invasion by forest pathogens in Europe. *New Phytologist, 197*(1), 238–250. https://doi.org/10.1111/j.1469-8137.2012.04364.x

Santos, B. A., Peres, C. A., Oliveira, M. A., Grillo, A., Alves-Costa, C. P., & Tabarelli, M. (2008). Drastic erosion in functional attributes of tree assemblages in Atlantic forest fragments of northeastern Brazil. *Biological Conservation, 141*(1), 249–260. https://doi.org/10.1016/j.biocon.2007.09.018

Santos, G. M., Aguiar, C. M. L., Genini, J., Martins, C. F., Zanella, F. C. V., & Mello, M. A. R. (2012). Invasive Africanized honeybees change the structure of native pollination networks in Brazil. *Biological Invasions, 14*(11), 2369–2378. https://doi.org/10.1007/s10530-012-0235-8

Sarà, G., Porporato, E. M. D., Mangano, M. C., & Mieszkowska, N. (2018). Multiple stressors facilitate the spread of a non-indigenous bivalve in the Mediterranean Sea. *Journal of Biogeography, 45*(5), 1090–1103. https://doi.org/10.1111/jbi.13184

Saunders, G., Cooke, B., McColl, K., Shine, R., & Peacock, T. (2010). Modern approaches for the biological control of vertebrate pests: An Australian perspective. *Biological Control, 52*(3), 288–295. https://doi.org/10.1016/j.biocontrol.2009.06.014

Scanes, E., Johnston, E., Cole, V., O'Connor, W., Parker, L., & Ross, P. (2016). Quantifying abundance and distribution of native and invasive oysters in an urbanised estuary. *Aquatic Invasions, 11*(4), 425–436. https://doi.org/10.3391/ai.2016.11.4.07

Schlaepfer, M. A., Sax, D. F., & Olden, J. D. (2011). The potential conservation value of non-native species. *Conservation Biology, 25*(3), 428–437. https://doi.org/10.1111/j.1523-1739.2010.01646.x

Schloegel, L. M., Toledo, L. F., Longcore, J. E., Greenspan, S. E., Vieira, C. A., Lee, M., Zhao, S., Wangen, C., Ferreira, C. M., Hipolito, M., Davies, A. J., Cuomo, C. A., Daszak, P., & James, T. Y. (2012). Novel, panzootic and hybrid genotypes of amphibian chytridiomycosis associated with the bullfrog trade. *Molecular Ecology, 21*(21), 5162–5177. https://doi.org/10.1111/j.1365-294X.2012.05710.x

Schmitz, C., Kreidenweis, U., Lotze-Campen, H., Popp, A., Krause, M., Dietrich, J. P., & Mueller, C. (2015). Agricultural trade and tropical deforestation: Interactions and related policy options. *Regional Environmental Change, 15*, 1757–1772. https://doi.org/10.1007/s10113-014-0700-2

Schnitzler, A., & Essl, F. (2015). From horticulture and biofuel to invasion: The spread of *Miscanthus* taxa in the USA and Europe. *Weed Research, 55*(3), Article 3. https://doi.org/10.1111/wre.12141

Schrama, M., & Bardgett, R. D. (2016). Grassland invasibility varies with drought effects on soil functioning. *Journal of Ecology, 104*(5), 1250–1258. https://doi.org/10.1111/1365-2745.12606

Schweiger, O., Biesmeijer, J. C., Bommarco, R., Hickler, T., Hulme, P. E., Klotz, S., Kühn, I., Moora, M., Nielsen, A., Ohlemüller, R., Petanidou, T., Potts, S. G., Pyšek, P., Stout, J. C., Sykes, M. T., Tscheulin, T., Vilà, M., Walther, G.-R., Westphal, C., … Settele, J. (2010). Multiple stressors on biotic interactions: How climate change and alien species interact to affect pollination. *Biological Reviews*, no-no. https://doi.org/10.1111/j.1469-185X.2010.00125.x

Schwilch, G., Adhikari, A., Jaboyedoff, M., Jaquet, S., Kaenzig, R., Liniger, H., Penna, I. M., Sudmeier-Rieux, K., & Upreti, B. R. (2017). Impacts of Outmigration on Land Management in a Nepali Mountain Area. In K. Sudmeier-Rieux, M. Fernández, I. M. Penna, M. Jaboyedoff, & J. C. Gaillard (Eds.), *Identifying Emerging Issues in Disaster Risk Reduction, Migration, Climate Change and Sustainable Development* (pp. 177–194). Springer International Publishing. https://doi.org/10.1007/978-3-319-33880-4_11

Scoleri, V. P., Johnson, C. N., Vertigan, P., & Jones, M. E. (2020). Conservation trade-offs: Island introduction of a threatened predator suppresses invasive mesopredators but eliminates a seabird colony. *Biological Conservation, 248*, 108635. https://doi.org/10.1016/j.biocon.2020.108635

Seabloom, E. W., Borer, E. T., Buckley, Y., Cleland, E. E., Davies, K., Firn, J., Harpole, W. S., Hautier, Y., Lind, E., MacDougall, A., Orrock, J. L., Prober, S. M., Adler, P., Alberti, J., Anderson, T. M., Bakker, J. D., Biederman, L. A., Blumenthal, D., Brown, C. S., … Yang, L. (2013). Predicting invasion in grassland ecosystems: Is exotic dominance the real embarrassment of richness? *Global Change Biology, 19*(12), Article 12. https://doi.org/10.1111/gcb.12370

Seastedt, T. R., Hobbs, R. J., & Suding, K. N. (2008). Management of novel ecosystems: Are novel approaches required? *Frontiers in Ecology and the Environment, 6*(10), 547–553. https://doi.org/10.1890/070046

Seddon, P. (2010). From Reintroduction to Assisted Colonization: Moving along the Conservation Translocation Spectrum. *Restoration Ecology, 18*(6), 796–802. https://doi.org/10.1111/j.1526-100X.2010.00724.x

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., van Kleunen, M., Winter, M., Ansong, M., Arianoutsou, M., Bacher, S., Blasius, B., Brockerhoff, E. G., Brundu, G., Capinha, C., Causton, C. E., Celesti-Grapow, L., … Essl, F. (2018). Global rise in emerging alien species results from increased accessibility of new source pools. *Proceedings of the National Academy of Sciences, 115*(10), E2264–E2273. https://doi.org/10.1073/pnas.1719429115

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., … Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications, 8*(1), 14435. https://doi.org/10.1038/ncomms14435

Seebens, H., Essl, F., Dawson, W., Fuentes, N., Moser, D., Pergl, J., Pyšek, P., Van Kleunen, M., Weber, E., Winter, M., & Blasius, B. (2015). Global trade will accelerate plant invasions in emerging economies under climate change. *Global Change Biology, 21*(11), 4128–4140. https://doi.org/10.1111/gcb.13021

Seebens, H., Gastner, M. T., & Blasius, B. (2013). The risk of marine bioinvasion caused by global shipping. *Ecology Letters, 16*(6), 782–790. https://doi.org/10.1111/ele.12111

Seebens, H., Schwartz, N., Schupp, P. J., & Blasius, B. (2016). Predicting the spread of marine species introduced by global shipping. *Proceedings of the National Academy of Sciences, 113*(20), 5646–5651. https://doi.org/10.1073/pnas.1524427113

Seipel, T., Kueffer, C., Rew, L. J., Daehler, C. C., Pauchard, A., Naylor, B. J., Alexander, J. M., Edwards, P. J., Parks, C. G., Arevalo, J. R., Cavieres, L. A., Dietz, H., Jakobs, G., McDougall, K., Otto, R., & Walsh, N. (2012). Processes at multiple scales affect richness and similarity of non-native plant species in mountains around the world. *Global Ecology and Biogeography, 21*(2), 236–246. https://doi.org/10.1111/j.1466-8238.2011.00664.x

Senasica. (2019). *PALOMILLA DEL NOPAL. Cactoblastis cactorum. Aviso público del riesgo y situación actual.* https://prod.senasica.gob.mx/SIRVEF/ContenidoPublico/Avisos%20y%20alertas/Avisos%20publicos/Aviso%20p%C3%BAblico%20Palomilla%20del%20nopal.pdf

Serbesoff-King, K. (2003). Melaleuca in Florida: A Literature Review on the Taxonomy, Distribution, Biology, Ecology, Economic Importance and Control Measures. *Journal of Aquatic Plant Management, 41,* 15.

Sestrich, C. M., McMahon, T. E., & Young, M. K. (2011). Influence of Fire on Native and Nonnative Salmonid Populations and Habitat in a Western Montana Basin. *Transactions of the American Fisheries Society, 140*(1), 136–146. https://doi.org/10.1080/00028487.2011.557019

Shackleton, C. M., & Gambiza, J. (2008). Social and ecological trade offs in combating land degradation: The case of invasion by a woody shrub *(Euryops Floribundus)* at Macubeni, South Africa. *Land Degradation & Development, 19*(4), 454–464. https://doi.org/10.1002/ldr.849

Shackleton, C. M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S. E., & Fabricius, C. (2007). Assessing the Effects of Invasive Alien Species on Rural Livelihoods: Case Examples and a Framework from South Africa. *Human Ecology, 35*(1), 113–127. https://doi.org/10.1007/s10745-006-9095-0

Shackleton, R. T., Richardson, D. M., Shackleton, C. M., Bennett, B., Crowley, S. L., Dehnen-Schmutz, K., Estévez, R. A., Fischer, A., Kueffer, C., Kull, C. A., Marchante, E., Novoa, A., Potgieter, L. J., Vaas, J., Vaz, A. S., & Larson, B. M. H. (2019). Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management, 229*, 10–26. https://doi.org/10.1016/j.jenvman.2018.04.045

Shackleton, R. T., Shackleton, C. M., & Kull, C. A. (2019). The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management, 229*, 145–157. https://doi.org/10.1016/j.jenvman.2018.05.007

Shackleton, S. E., & Shackleton, R. T. (2018). Local knowledge regarding ecosystem services and disservices from invasive alien plants in the arid Kalahari, South Africa. *Journal of Arid Environments, 159*, 22–33. https://doi.org/10.1016/j.jaridenv.2017.07.001

Sharma, A., Aggarwal, N. K., Saini, A., & Yadav, A. (2015). Beyond Biocontrol: Water Hyacinth-Opportunities and Challenges. *Journal of Environmental Science and Technology, 9*(1), 26–48. https://doi.org/10.3923/jest.2016.26.48

Sharrock, S. L. (2011). The biodiversity benefits of botanic gardens. *Trends in Ecology & Evolution, 26*(9), 433. https://doi.org/10.1016/j.tree.2011.05.008

Shearer, B. L., Crane, C. E., Barrett, S., & Cochrane, A. (2007). *Phytophthora cinnamomi* invasion, a major threatening process to conservation of flora diversity in the South-west Botanical Province of Western Australia. *Australian Journal of Botany, 55*(3), 225–238. https://doi.org/10.1071/BT06019

Shelton, W. L., & Rothbard, S. (2006). Exotic species in global aquaculture—A review. *Israeli Journal of Aquaculture – Bamidgeh, 58*(1), 3–28. https://doi.org/10.46989/001c.20429

Shiba-Reyes, M., Troyo, E., Martínez-Rincón, R., & Breceda, A. (2021). Impacts of tropical hurricanes on the vegetation cover of the lower basin and estuary of San José del Cabo, Baja California Sur, Mexico. *Revista Chapingo Serie Ciencias Forestales y Del Ambiente, 27*(1), 165–180. http://dx.doi.org/10.5154/r.rchscfa.2020.03.011

Shimada, N. (2016). Outline of the Great East Japan Earthquake. In *Ecological Impacts of Tsunamis on Coastal Ecosystems* (pp. 1–8). Springer. https://link.springer.com/chapter/10.1007/978-4-431-56448-5_1

Shine, R. (2010). The Ecological Impact of Invasive Cane Toads *(Bufo marinus)* in Australia. *The Quarterly Review of Biology, 85*(3), Article 3. https://doi.org/10.1086/655116

Shochat, E., Lerman, S. B., Anderies, J. M., Warren, P. S., Faeth, S. H., & Nilon, C. H. (2010). Invasion, Competition, and Biodiversity Loss in Urban Ecosystems. *BioScience, 60*(3), 199–208. https://doi.org/10.1525/bio.2010.60.3.6

Shukla, P. R., Skea, J., Calvo Buendia, E., Masson-Delmotte, V., Pörtner, H. O., Roberts, D. C., Zhai, P., Slade, R., Connors, S., & Van Diemen, R. (2019). *Climate Change and Land: An IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems.* Intergovernmental Panel on Climate Change (IPCC). https://www.ipcc.ch/srccl/

Shwartz, A., Strubbe, D., Butler, C. J., Matthysen, E., & Kark, S. (2009). The effect of enemy-release and climate conditions on invasive birds: A regional test using the rose-ringed parakeet *(Psittacula krameri)* as a case study. *Diversity and Distributions,* *15*(2), 310–318. https://doi.org/10.1111/j.1472-4642.2008.00538.x

Silcock, J. L. (2018). Aboriginal Translocations: The Intentional Propagation and Dispersal of Plants in Aboriginal Australia. *Journal of Ethnobiology, 38*(3), 390. https://doi.org/10.2993/0278-0771-38.3.390

Simberloff, D. (2006). Invasional meltdown 6 years later: Important phenomenon, unfortunate metaphor, or both? *Ecology Letters, 9*(8), 912–919. https://doi.org/10.1111/j.1461-0248.2006.00939.x

Simberloff, D., & Rejmanek, M. (Eds.). (2011). *Encyclopedia of Biological Invasions* (Vol. 3). University of California Press. https://doi.org/10.1525/9780520948433

Simberloff, D., & Von Holle, B. (1999). Positive Interactions of Nonindigenous Species: Invasional Meltdown? *Biological Invasions, 1*(1), 21–32. https://doi.org/10.1023/A:1010086329619

Simkanin, C., Carlton, J. T., Steves, B., Fofonoff, P., Nelson, J. C., Clarke Murray, C., & Ruiz, G. M. (2019). Exploring potential establishment of marine rafting species after transoceanic long-distance dispersal. *Global Ecology and Biogeography, 28*(5), 588–600. https://doi.org/10.1111/geb.12878

Simkanin, C., Davidson, I. C., Dower, J. F., Jamieson, G., & Therriault, T. W. (2012). Anthropogenic structures and the infiltration of natural benthos by invasive ascidians. *Marine Ecology, 33*(4), 499–511. https://doi.org/10.1111/j.1439-0485.2012.00516.x

Simler, A. B., Williamson, M., Schwartz, M. W., & Rizzo, D. M. (2019). Amplifying plant disease risk through assisted migration. *Conservation Letters, 12*(2), e12605. https://doi.org/10.1111/conl.12605

Simões, K. C. C., Hay, J. D. V., Andrade, C. O. de, Júnior, O. A. de C., & Gomes, R. A. T. (2014). Distribuição de cana-do-reino (*Arundo donax* L.) no Distrito Federal, Brasil. *Biodiversidade Brasileira, 0*(2), Article 2. https://www.icmbio.gov.br/revistaeletronica/index.php/BioBR/article/view/329

Simon, R., Xie, C. H., Clausen, A., Jansky, S. H., Halterman, D., Conner, T., Knapp, S., Brundage, J., Symon, D., & Spooner, D. (2010). Wild and Cultivated Potato (*Solanum* sect. *Petota*) Escaped and Persistent Outside of its Natural Range. *Invasive Plant Science and Management, 3*(3), 286–293. https://doi.org/10.1614/IPSM-D-09-00043.1

Singh, R. S. (1985). *Diseases of Vegetable Crops.* Oxford & IBH Publishers.

Sloane, D. R., Ens, E., Wunungmurra, Y., Gumana, Y., Wunungmurra, B., Wirrpanda, M., Towler, G., Preece, D., & Rangers, Y. (2021). Lessons from old fenced plots: Eco-cultural Impacts of feral ungulates and potential decline in sea-level rise resilience of coastal floodplains in northern Australia. *Ecological Management & Restoration, 22*(2), 191–203. https://doi.org/10.1111/emr.12464

Smith, E. R. C., Bennion, H., Sayer, C. D., Aldridge, D. C., & Owen, M. (2020). Recreational angling as a pathway for invasive non-native species spread: Awareness of biosecurity and the risk of long distance movement into Great Britain. *Biological Invasions, 22*(3), 1135–1159. https://doi.org/10.1007/s10530-019-02169-5

Smith, H. T., & Engeman, R. M. (2003). A review of the range, distribution, and ecology of the invasive northern curly-tailed lizard in Florida. *USDA National Wildlife Research Center-Staff Publications*, 218. https://doi.org/10.1111/emr.12464

Smith, L. L., Allen, D. J., & Barney, J. (2015). The thin green line: Sustainable bioenergy feedstocks or invaders in waiting. *NeoBiota, 25*, 47.

Smith, N. H. (2012). The global distribution and phylogeography of *Mycobacterium bovis* clonal complexes. *Infection, Genetics and Evolution, 12*(4), Article 4. https://doi.org/10.1016/j.meegid.2011.09.007

Smith, P. T., Imbun, B. Y., & Duarte, F. P. (2016). Impacts of a Fish Kill at Lake Kutubu, Papua New Guinea. *Pacific Science, 70*(1), 21–33. https://doi.org/10.2984/70.1.2

Smith, S. D., Huxman, T. E., Zitzer, S. F., Charlet, T. N., Housman, D. C., Coleman, J. S., Fenstermaker, L. K., Seemann, J. R., & Nowak, R. S. (2000). Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. *Nature, 408*(6808), 79–82. https://doi.org/10.1038/35040544

Smyser, T. J., Tabak, M. A., Slootmaker, C., Robeson, M. S., Miller, R. S., Bosse, M., Megens, H.-J., Groenen, M. A. M., Paiva, S. R., Faria, D. A. de, Blackburn, H. D., Schmit, B. S., & Piaggio, A. J. (2020). Mixed ancestry from wild and domestic lineages contributes to the rapid expansion of invasive feral swine. *Molecular Ecology, 29*(6), 1103–1119. https://doi.org/10.1111/mec.15392

Snyder, M. R., Stepien, C. A., Marshall, N. T., Scheppler, H. B., Black, C. L., & Czajkowski, K. P. (2020). Detecting aquatic invasive species in bait and pond stores with targeted environmental (e)DNA high-throughput sequencing metabarcode assays: Angler, retailer, and manager implications. *Biological Conservation, 245*, 108430. https://doi.org/10.1016/j.biocon.2020.108430

Snyder, W. E., & Evans, E. W. (2006). Ecological Effects of Invasive Arthropod Generalist Predators. *Annual Review of Ecology, Evolution, and Systematics, 37*(1), 95–122. https://doi.org/10.1146/annurev.ecolsys.37.091305.110107

Sodhi, N. S., Koh, L. P., Clements, R., Wanger, T. C., Hill, J. K., Hamer, K. C., Clough, Y., Tscharntke, T., Posa, M. R. C., & Lee, T. M. (2010). Conserving Southeast Asian forest biodiversity in human-modified landscapes. *Biological Conservation, 143*(10), 2375–2384. https://doi.org/10.1016/j.biocon.2009.12.029

Solberg, M. F., Robertsen, G., Sundt-Hansen, L. E., Hindar, K., & Glover, K. A. (2020). Domestication leads to increased predation susceptibility. *Scientific Reports, 10*(1), 1929. https://doi.org/10.1038/s41598-020-58661-9

Somarakis, S., Pavlidis, M., Saapoglou, C., Tsigenopoulos, C. S., & Dempster, T. (2013). Evidence for 'escape through spawning'in large gilthead sea bream *Sparus aurata* reared in commercial sea-cages. *Aquaculture Environment Interactions, 3*(2), 135–152. https://doi.org/10.3354/aei00057

Sonter, L. J., Moran, C. J., Barrett, D. J., & Soares-Filho, B. S. (2014). Processes of land use change in mining regions. *Journal of Cleaner Production, 84*, 494–501. https://doi.org/10.1016/j.jclepro.2014.03.084

Soon, J. M., & Manning, L. (2018). Food smuggling and trafficking: The key factors of influence. *Trends in Food Science & Technology, 81*, 132–138. https://doi.org/10.1016/j.tifs.2018.09.007

Souza-Alonso, P., Rodríguez, J., González, L., & Lorenzo, P. (2017). Here to stay. Recent advances and perspectives about *Acacia* invasion in Mediterranean areas. *Annals of Forest Science, 74*(3), 55. https://doi.org/10.1007/s13595-017-0651-0

Space, J. C., & Flynn, T. (2001). *Report to the Kingdom of Tonga on Invasive Plant Species of Environmental Concern* (p. 79). U.S.D.A. Forest Service, Pacific Southwest Research Station Institute of Pacific Islands Forestry Honolulu, Hawai'i, USA. http://hear.org/pier/pdf/tonga_report.pdf

Spear, D., & Chown, S. L. (2009). The extent and impacts of ungulate translocations: South Africa in a global context. *Biological Conservation, 142*(2), 353–363. https://doi.org/10.1016/j.biocon.2008.10.031

Spear, D., Foxcroft, L. C., Bezuidenhout, H., & McGeoch, M. A. (2013). Human population density explains alien species richness in protected areas. *Biological Conservation, 159*, 137–147. https://doi.org/10.1016/j.biocon.2012.11.022

Spooner, P. G. (2015). Minor rural road networks: Values, challenges, and opportunities for biodiversity conservation. *Nature Conservation, 11*, 129–142. https://doi.org/10.3897/natureconservation.11.4434

Squires, V. R., & Glenn, E. P. (2011). Salination, Desertification, and Soil Erosion. *EOLSS Publications, 3*, 9. https://www.eolss.net/sample-chapters/C10/E5-01A-04-05.pdf

Sritrairat, S., Peteet, D. M., Kenna, T. C., Sambrotto, R., Kurdyla, D., & Guilderson, T. (2012). A history of vegetation, sediment and nutrient dynamics at Tivoli North Bay, Hudson Estuary, New York. *Estuarine, Coastal and Shelf Science, 102–103*, 24–35. https://doi.org/10.1016/j.ecss.2012.03.003

St. Clair, S. B., O'Connor, R., Gill, R., & McMillan, B. (2016). Biotic resistance and disturbance: Rodent consumers regulate post-fire plant invasions and increase plant community diversity. *Ecology, 97*(7), 1700–1711. https://doi.org/10.1002/ecy.1391

Steffen, W., Broadgate, W., Deutsch, L., Gaffney, O., & Ludwig, C. (2015). The trajectory of the Anthropocene: The Great Acceleration. *The Anthropocene Review, 2*(1), 81–98. https://doi.org/10.1177/2053019614564785

Stentiford, G. D., & Lightner, D. V. (2011). Cases of White Spot Disease (WSD) in European shrimp farms. *Aquaculture, 319*(1), 302–306. https://doi.org/10.1016/j.aquaculture.2011.06.032

Stephens, A. E. A., Kriticos, D. J., & Leriche, A. (2007). The current and future potential geographical distribution of the oriental fruit fly, *Bactrocera dorsalis* (Diptera: Tephritidae). *Bulletin of Entomological Research, 97*(4), 369–378. https://doi.org/10.1017/S0007485307005044

Stevens, C. J. (2019). Nitrogen in the environment. *Science, 363*(6427), 578–580. https://doi.org/10.1126/science.aav8215

Stilley, J. A., & Gabler, C. A. (2021). Effects of Patch Size, Fragmentation, and Invasive Species on Plant and Lepidoptera Communities in Southern Texas. *Insects, 12*(9), 777. https://doi.org/10.3390/insects12090777

Stuart-Smith, R. D., Edgar, G. J., Stuart-Smith, J. F., Barrett, N. S., Fowles, A. E., Hill, N. A., Cooper, A. T., Myers, A. P., Oh, E. S., Pocklington, J. B., & Thomson, R. J. (2015). Loss of native rocky reef biodiversity in Australian metropolitan embayments. *Marine Pollution Bulletin, 95*(1), 324–332. https://doi.org/10.1016/j.marpolbul.2015.03.023

Suárez-Esteban, A., Fahrig, L., Delibes, M., & Fedriani, J. M. (2016). Can anthropogenic linear gaps increase plant abundance and diversity? *Landscape Ecology, 31*(4), 721–729. https://doi.org/10.1007/s10980-015-0329-7

Subbarao, K. V., Sundin, G. W., & Klosterman, S. J. (2015). Focus Issue Articles on Emerging and Re-Emerging Plant Diseases. *Phytopathology™, 105*(7), 852–854. https://doi.org/10.1094/PHYTO-105-7-0001

Suckling, D. M., & Sforza, R. F. H. (2014). What Magnitude Are Observed Non-Target Impacts from Weed Biocontrol? *PLoS ONE, 9*(1), e84847. https://doi.org/10.1371/journal.pone.0084847

Sullivan, J. J., Williams, P. A., Timmins, S. M., & Smale, M. C. (2009). Distribution and spread of environmental weeds along New Zealand roadsides. *New Zealand Journal of Ecology, 33*(2), 190–204. https://researcharchive.lincoln.ac.nz/handle/10182/4962

Svenning, J.-C., & Sandel, B. (2013). Disequilibrium Vegetation Dynamics Under Future Climate Change. *American Journal of Botany, 100*(7), 1266–1286. https://doi.org/10.3732/ajb.1200469

Tabarelli, M., Lopes, A. V., & Peres, C. A. (2008). Edge-effects Drive Tropical Forest Fragments Towards an Early-Successional System. *Biotropica, 40*(6), 657–661. https://doi.org/10.1111/j.1744-7429.2008.00454.x

Tamburello, L., Bulleri, F., Balata, D., & Benedetti-Cecchi, L. (2014). The role of overgrazing and anthropogenic disturbance in shaping spatial patterns of distribution of an invasive seaweed. *Journal of Applied Ecology, 51*(2), 406–414. https://doi.org/10.1111/1365-2664.12199

Tano, S. A., Halling, C., Lind, E., Buriyo, A., & Wikström, S. A. (2015). Extensive spread of farmed seaweeds causes a shift from native to non-native haplotypes in natural seaweed beds. *Marine Biology, 162*(10), 1983–1992. https://doi.org/10.1007/s00227-015-2724-7

Tatem, A. J. (2009). The worldwide airline network and the dispersal of exotic species: 2007-2010. *Ecography, 32*(1), 94–102. https://doi.org/10.1111/j.1600-0587.2008.05588.x

Taylor, S., Kumar, L., & Reid, N. (2012). Impacts of climate change and land-use on the potential distribution of an invasive weed: A case study of *Lantana camara* in Australia. *Weed Research, 52*(5), 391–401. https://doi.org/10.1111/j.1365-3180.2012.00930.x

Tedeschi, L., Biancolini, D., Capinha, C., Rondinini, C., & Essl, F. (2021). Introduction, spread, and impacts of invasive alien mammal species in Europe. *Mammal Review*, mam.12277. https://doi.org/10.1111/mam.12277

Teletchea, F. (2019). Fish domestication in aquaculture: Reassessment and emerging questions. *Cybium, 43*(1), 7–15. https://doi.org/10.26028/cybium/2019-431-001

Téllez, T. R., López, E., Granado, G., Pérez, E., López, R., & Guzmán, J. (2008). The Water Hyacinth, *Eichhornia crassipes*: An invasive plant in the Guadiana River Basin (Spain). *Aquatic Invasions, 3*(1), 42–53. https://doi.org/10.3391/ai.2008.3.1.8

Thaler, A. D., Freitag, A., Bergman, E., Fretz, D., & Saleu, W. (2015). Robots as Vectors for Marine Invasions: Best Practices for Minimizing Transmission of Invasive Species *Via* Observation-Class ROVs. *Tropical Conservation Science, 8*(3), 711–717. https://doi.org/10.1177/194008291500800308

Thamaga, K. H., & Dube, T. (2018). Remote sensing of invasive water hyacinth *(Eichhornia crassipes)*: A review on applications and challenges. *Remote Sensing Applications: Society and Environment, 10*, 36–46. https://doi.org/10.1016/j.rsase.2018.02.005

Theoharides, K. A., & Dukes, J. S. (2007). Plant invasion across space and time: Factors affecting nonindigenous species success during four stages of invasion. *New Phytologist, 176*(2), 256–273. https://doi.org/10.1111/j.1469-8137.2007.02207.x

Thia, J. P. (2016). Trade and Urbanisation. *World Economy, 39*, 853–872. https://doi.org/10.1111/twec.13187

Thiengo, S. C., Faraco, F. A., Salgado, N. C., Cowie, R. H., & Fernandez, M. A. (2007). Rapid spread of an invasive snail in South America: The giant African snail, *Achatina fulica*, in Brasil. *Biological Invasions, 9*(6), 693–702. https://doi.org/10.1007/s10530-006-9069-6

Thomas, Z. A., Turney, C. S. M., Palmer, J. G., Lloydd, S., Klaricich, J. N. L., & Hogg, A. (2018). Extending the observational record to provide new insights into invasive alien species in a coastal dune environment of New Zealand. *Applied Geography, 98*, 100–109. https://doi.org/10.1016/j.apgeog.2018.07.006

Thompson, J., Lugo, A. E., & Thomlinson, J. (2007). Land use history, hurricane disturbance, and the fate of introduced species in a subtropical wet forest in Puerto Rico. *Plant Ecology, 192*, 289–301. https://doi.org/10.1007/s11258-007-9318-5

Timbilla, J. A., & Braimah, H. (1993). A survey of the introduction, distribution and spread of *Chromolaena odorata* in Ghana. *Distribution, Ecology and Management of Chromolaena Odorata, 202*, 6–18. https://horizon.documentation.ird.fr/exl-doc/pleins_textes/doc34-03/010006224.pdf

Todd, P. A., Heery, E. C., Loke, L. H. L., Thurstan, R. H., Kotze, D. J., & Swan, C. (2019). Towards an urban marine ecology: Characterizing the drivers, patterns and processes of marine ecosystems in coastal cities. *Oikos, 128*(9), 1215–1242. https://doi.org/10.1111/oik.05946

Toepfer, I., Favet, J., Schulte, A., Schmölling, M., Butte, W., Triplett, E. W., & Broughton, W. J. (2012). Pathogens as potential hitchhikers on intercontinental dust. *Aerobiologia, 28*(2), 221–231. https://doi.org/10.1007/s10453-011-9230-2

Tolfts, A. (1997). Cordia alliodora: The best laid plans. *Alien, 6*, 12–13.

Tollington, S., Turbé, A., Rabitsch, W., Groombridge, J. J., Scalera, R., Essl, F., & Shwartz, A. (2017). Making the EU Legislation on Invasive Species a Conservation Success. *Conservation Letters, 10*(1), 112–120. https://doi.org/10.1111/conl.12214

Tooth, I. M., & Leishman, M. R. (2014). Elevated carbon dioxide and fire reduce biomass of native grass species when grown in competition with invasive exotic grasses in a savanna experimental system. *Biological Invasions, 16*(2), 257–268. https://doi.org/10.1007/s10530-013-0448-5

Tøttrup, A. P., Chan, B. K. K., Koskinen, H., & Høeg, J. T. (2010). 'Flying barnacles': Implications for the spread of non-indigenous species. *Biofouling, 26*(5), 577–582. https://doi.org/10.1080/08927014.2010.489203

Touza, J., Dehnen-Schmutz, K., & Jones, G. (2008). Economic Analysis of Invasive Species Policies. *Biological Invasions, 193*, 353–366. https://doi.org/10.1007/978-3-540-36920-2_20

Towerton, A. L., Penman, T. D., Kavanagh, R. P., & Dickman, C. R. (2011). Detecting pest and prey responses to fox control across the landscape using remote cameras. *Wildlife Research, 38*(3), 208. https://doi.org/10.1071/WR10213

Traill, L. W., Perhans, K., Lovelock, C. E., Prohaska, A., McFallan, S., Rhodes, J. R., & Wilson, K. A. (2011). Managing for Change: Wetland Transitions under Sea-Level Rise and Outcomes for Threatened Species. *Diversity and Distributions, 17*(6), 1225–1233. https://doi.org/10.1111/j.1472-4642.2011.00807.x

Traveset, A., & Richardson, D. M. (2014). Mutualistic Interactions and Biological Invasions. *Annual Review of Ecology, Evolution, and Systematics, 45*(1), 89–113. https://doi.org/10.1146/annurev-ecolsys-120213-091857

Tricarico, E. (2012). A review on pathways and drivers of use regarding non-native freshwater fish introductions in the Mediterranean region. *Fisheries Management and Ecology, 19*(2), 133–141. https://doi.org/10.1111/j.1365-2400.2011.00834.x

Tripathi, S. (2020). Do macroeconomic factors promote urbanization? Evidence from BRICS countries. *Asia-Pacific Journal of Regional Science, 5*, 397–426. https://doi.org/10.1007/s41685-020-00179-z

Trombulak, S. C., & Frissell, C. A. (2000). Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities. *Conservation Biology, 14*(1), 18–30. https://doi.org/10.1046/j.1523-1739.2000.99084.x

Truscott, A.-M., Soulsby, C., Palmer, S. C. F., Newell, L., & Hulme, P. E. (2006). The dispersal characteristics of the invasive plant *Mimulus guttatus* and the ecological significance of increased occurrence of high-flow events. *Journal of Ecology, 94*(6), 1080–1091. https://doi.org/10.1111/j.1365-2745.2006.01171.x

Tuinoula, M. (2003). American Samoa. In C. Shine, J. K. Reaser, & A. T. Gutierrez (Eds.), *Prevention and Management of Invasive Alien Species: Proceedings of a Workshop on Forging Cooperation Throughout the Austral-Pacific* (pp. 64–65). https://www.doi.gov/sites/doi.gov/files/uploads/forging_cooperation_in_the_austral_pacific.pdf

Turbelin, A. J., Malamud, B. D., & Francis, R. A. (2017). Mapping the global state of invasive alien species: Patterns of invasion and policy responses. *Global Ecology and Biogeography, 26*(1), 78–92. https://doi.org/10.1111/geb.12517

Tylianakis, J. M., Didham, R. K., Bascompte, J., & Wardle, D. A. (2008). Global change and species interactions in terrestrial ecosystems. *Ecology Letters, 11*(12), 1351–1363. https://doi.org/10.1111/j.1461-0248.2008.01250.x

UNEP. (2019). *Global Environment Outlook-GEO-6: Healthy Planet, Healthy People.* Cambridge University Press. https://doi.org/10.1017/9781108627146

UNHCR. (2020). UNHCR Global Report 2020. *UNHCR Flagship Reports.* https://www.unhcr.org/flagship-reports/globalreport/

United Nations. (1997). *Kyoto Protocol to the United Nations Framework Convention on Climate Change.* https://unfccc.int/sites/default/files/resource/docs/cop3/l07a01.pdf

United Nations. (2020). UN World Water Development Report 2020. *UN-Water*. https://www.unwater.org/publications/world-water-development-report-2020/

United Nations Convention to Combat Desertification. (2017). *Global Land Outlook.*

United Nations Department of Economic and Social Affairs. (2019). *World Urbanization Prospects 2018: Highlights.* https://doi.org/10.18356/6255ead2-en

United Nations, Department of Economic and Social Affairs, & Population Division. (2019). *World population prospects Highlights, 2019 revision Highlights, 2019 revision.*

United Nations Department of Economic and Social Affairs, Population Division. (2020). *International Migration 2020 Highlights (ST/ESA/SER.A/452).* https://www.un.org/en/desa/international-migration-2020-highlights

United Nations Environment Programme. (2021). *From Pollution to Solution. A global assessment of marine litter and plastic pollution Nairobi.* https://nottingham-repository.worktribe.com/output/8047115

UNWTO. (2021). *Compendium of Tourism Statistics Data 2015 – 2019.* https://www.e-unwto.org/doi/epdf/10.18111/9789284422494.1

Uyi, O. O., & Igbinosa, I. B. (2013). The status of *Chromolaena odorata* and its biocontrol in West Africa. *Proceedings of the Eighth International Workshop on Biological Control and Management of Chromolaena Odorata and Other Eupatorieae, 1-2 November 2010, Nairobi, Kenya,* 86–98. https://www.cabdirect.org/cabdirect/abstract/20173115399

Uzair, M., Ahmed, M., & Nazim, K. (2009). Effect of industrial waste on seed bank and growth of wild plants in Dhabeji area, Karachi, Pakistan. *Pakistan Journal of Botany, 41*(4), 1659–1665. http://pakbs.org/pjbot/PDFs/41(4)/PJB41(4)1659.pdf

van den Burg, M. P., Brisbane, J. L. K., & Knapp, C. R. (2019). Post-hurricane relief facilitates invasion and establishment of two invasive alien vertebrate species in the Commonwealth of Dominica, West Indies. *Biological Invasions.* https://doi.org/10.1007/s10530-019-02107-5

van der Zanden, E. H., Verburg, P. H., Schulp, C. J. E., & Verkerk, P. J. (2017). Trade-offs of European agricultural abandonment. *Land Use Policy, 62*, 290–301. https://doi.org/10.1016/j.landusepol.2017.01.003

van Kleunen, M., Essl, F., Pergl, J., Brundu, G., Carboni, M., Dullinger, S., Early, R., González-Moreno, P., Groom, Q. J., Hulme, P. E., Kueffer, C., Kühn, I., Máguas, C., Maurel, N., Novoa, A., Parepa, M., Pyšek, P., Seebens, H., Tanner, R., … Dehnen-Schmutz, K. (2018). The changing role of ornamental horticulture in alien plant invasions. *Biological Reviews, 93*(3), 1421–1437. https://doi.org/10.1111/brv.12402

van Kleunen, M., Xu, X., Yang, Q., Maurel, N., Zhang, Z., Dawson, W., Essl, F., Kreft, H., Pergl, J., Pyšek, P., Weigelt, P., Moser, D., Lenzner, B., & Fristoe, T. S. (2020). Economic use of plants is key to their naturalization success. *Nature Communications, 11*(1), 3201. https://doi.org/10.1038/s41467-020-16982-3

Van Leeuwen, C. H. (2018). Internal and external dispersal of plants by animals: An aquatic perspective on alien interference. *Frontiers in Plant Science, 9*, 153.

van Ruijven, J., De Deyn, G. B., & Berendse, F. (2003). Diversity reduces invasibility in experimental plant communities: The role of plant species. *Ecology Letters, 6*(10), 910–918. https://doi.org/10.1046/j.1461-0248.2003.00516.x

Van Wilgen, B. W., Dyer, C., Hoffmann, J. H., Ivey, P., Le Maitre, D. C., Moore, J. L., Richardson, D. M., Rouget, M., Wannenburgh, A., & Wilson, J. R. U. (2011). National-scale strategic approaches for managing introduced plants: Insights from Australian *acacias* in South Africa: Strategic approaches for managing introduced acacias. *Diversity and Distributions, 17*(5), 1060–1075. https://doi.org/10.1111/j.1472-4642.2011.00785.x

van Zwieten, P. A. M., Kolding, J., Plank, M. J., Hecky, R. E., Bridgeman, T. B., MacIntyre, S., Seehausen, O., & Silsbe, G. M. (2015). The Nile perch invasion in Lake Victoria: Cause or consequence of the haplochromine decline? *Canadian Journal of Fisheries and Aquatic Sciences, 73*(4), 622–643. https://doi.org/10.1139/cjfas-2015-0130

Vaselli, S., Bulleri, F., & Benedetti-Cecchi, L. (2008). Hard coastal-defence structures as habitats for native and exotic rocky-bottom species. *Marine Environmental Research, 66*(4), 395–403. https://doi.org/10.1016/j.marenvres.2008.06.002

Vasquez, E. A., Glenn, E. P., Brown, J. J., Guntenspergen, G. R., & Nelson, S. G. (2005). Salt tolerance underlies the cryptic invasion of North American salt marshes by an introduced haplotype of the common reed *Phragmites australis* (Poaceae). *Marine Ecology Progress Series, 298*, 1–8. https://doi.org/10.3354/meps298001

Vasquez, E. A., Sheley, R., & Svejcar, T. (2008). Creating Invasion Resistant Soils *via* Nitrogen Management. *Invasive Plant Science and Management, 1*(3), Article 3. https://doi.org/10.1614/IPSM-07-059.1

Vaverková, M. D., Adamcová, D., Winkler, J., Koda, E., Petrželová, L., & Maxianová, A. (2020). Alternative method of composting on a reclaimed municipal waste landfill in accordance with the circular economy: Benefits and risks. *Science of The Total Environment, 723*, 137971. https://doi.org/10.1016/j.scitotenv.2020.137971

Veiga, A. (2021, March 30). *Suez Canal blockage adds to pressure points in global trade.* AP NEWS. https://apnews.com/article/europe-global-trade-egypt-coronavirus-pandemic-suez-canal-166bc8f21e9705f2921a67ef2dea176c

Veldman, J. W., Mostacedo, B., Peña-Claros, M., & Putz, F. E. (2009). Selective logging and fire as drivers of alien grass invasion in a Bolivian tropical dry forest. *Forest Ecology and Management, 258*(7), 1643–1649. https://doi.org/10.1016/j.foreco.2009.07.024

Veldman, J. W., & Putz, F. E. (2010). Long-distance Dispersal of Invasive Grasses by Logging Vehicles in a Tropical Dry Forest. *Biotropica, 42*(6), 697–703. JSTOR. https://doi.org/10.1111/j.1744-7429.201 0.00647.x

Velle, L. G., Nilsen, L. S., Norderhaug, A., & Vandvik, V. (2014). Does prescribed burning result in biotic homogenization of coastal heathlands? *Global Change Biology, 20*(5), 1429–1440. https://doi.org/10.1111/gcb.12448

Verbrugge, L. N. H., Schipper, A. M., Huijbregts, M. A. J., Van der Velde, G., & Leuven, R. S. E. W. (2012). Sensitivity of native and non-native mollusc species to changing river water temperature and salinity. *Biological Invasions, 14*(6), 1187–1199. https://doi.org/10.1007/s10530-011-0148-y

Verlaque, M., Ruitton, S., Mineur, F., & Boudouresque, C.-F. (2015). *CIESM Atlas of Exotic species in the Mediterranean Sea*. https://www.ciesm.org/online/archives/abstracts/pdf/38/CIESM_Congress_2007_Istanbul_article_0014.pdf

Vermonden, K., Leuven, R. S. E. W., & van der Velde, G. (2010). Environmental factors determining invasibility of urban waters for exotic macroinvertebrates. *Diversity and Distributions, 16*(6), 1009–1021. https://doi.org/10.1111/j.1472-4642.2010.00702.x

Vestergaard, N., Kaiser, B., Fernandez, L., & Larsen, J. N. (Eds.). (2018). *Arctic Marine Resource Governance and Development.* Springer. https://doi.org/10.1007/978-3-319-67365-3

Vidal, E. M., Tjaden, J. D., & Global Migration Data Analysis Centre. (2018). *Global migration indicators 2018: Insights from the Global Migration* Data Portal: www.migrationdataportal.org. https://publications.iom.int/system/files/pdf/global_migration_indicators_2018.pdf

Vilà, M., & Ibáñez, I. (2011). Plant invasions in the landscape. *Landscape Ecology, 26*(4), Article 4. https://doi.org/10.1007/s10980-011-9585-3

Violle, C., Nemergut, D. R., Pu, Z., & Jiang, L. (2011). Phylogenetic limiting similarity and competitive exclusion. *Ecology Letters, 14*(8), 782–787. https://doi.org/10.1111/j.1461-0248.2011.01644.x

Voight, J. R., Lee, R. W., Reft, A. J., & Bates, A. E. (2012). Scientific gear as a vector for non-native species at deep-sea hydrothermal vents. *Conservation Biology, 26*(5), 938–942. https://doi.org/10.1111/j.1523-1739.2012.01864.x

Vranken, S., Bosch, S., Peña, V., Leliaert, F., Mineur, F., & De Clerck, O. (2018). A risk assessment of aquarium trade introductions of seaweed in European waters. *Biological Invasions, 20*(5), 1171–1187. https://doi.org/10.1007/s10530-017-1618-7

Vrees, L. de. (2021). Adaptive marine spatial planning in the Netherlands sector of the North Sea. *Marine Policy, 132*, 103418. https://doi.org/10.1016/j.marpol.2019.01.007

Waddell, E. H., Banin, L. F., Fleiss, S., Hill, J. K., Hughes, M., Jelling, A., Yeong, K. L., Ola, B. B., Sailim, A. B., Tangah, J., & Chapman, D. S. (2020). Land-use change and propagule pressure promote plant invasions in tropical rainforest remnants. *Landscape Ecology, 35*, 1891–1906. https://doi.org/10.1007/s10980-020-01067-9

Wagner, D. L., Metzler, K. J., Leicht-Young, S. A., & Motzkin, G. (2014). Vegetation composition along a New England transmission line corridor and its implications for other trophic levels. *Forest Ecology and Management, 327*, 231–239. https://doi.org/10.1016/j.foreco.2014.04.026

Wakie, T. T., Laituri, M., & Evangelista, P. H. (2016). Assessing the distribution and impacts of *Prosopis juliflora* through participatory approaches. *Applied Geography, 66*, 132–143. https://doi.org/10.1016/j.apgeog.2015.11.017

Walters, L. J., Brown, K. R., Stam, W. T., & Olsen, J. L. (2006). E-commerce and *Caulerpa*: Unregulated dispersal of invasive species. *Frontiers in Ecology and the Environment, 4*(2), 75–79. https://doi.org/10.1890/1540-9295(2006)004[0075:EACUDO]2.0.CO;2

Walther, G.-R., Roques, A., Hulme, P. E., Sykes, M. T., Pyšek, P., Kühn, I., Zobel, M., Bacher, S., Botta-Dukát, Z., Bugmann, H., Czúcz, B., Dauber, J., Hickler, T., Jarošík, V., Kenis, M., Klotz, S., Minchin, D., Moora, M., Nentwig, W., … Settele, J. (2009). Alien species in a warmer world: Risks and opportunities. *Trends in Ecology & Evolution, 24*(12), 686–693. https://doi.org/10.1016/j.tree.2009.06.008

Walton, C. S. (2003). *Leucaena* (Leucaena leucocephala) *in Queensland.* Land Protection, Department of Natural Resources, Mines, and Energy.

Walton, W. E., Henke, J. A., & Why, A. M. (2012). *Gambusia affinis* (Baird and Girard) and *Gambusia holbrooki* Girard (Mosquitofish). In *A handbook of global freshwater invasive species* (pp. 261–273). Earthscan London.

Wang, Q., Cheng, L., Liu, J., Li, Z., Xie, S., & De Silva, S. S. (2015). Freshwater aquaculture in PR China: Trends and prospects. *Reviews in Aquaculture, 7*(4), 283–302. https://doi.org/10.1111/raq.12086

Wang, Q., Fan, X., & Wang, M. (2016). Evidence of high-elevation amplification *versus* Arctic amplification. *Scientific Reports, 6*(1), 19219. https://doi.org/10.1038/srep19219

Wang, Q., Zhuang, Z., Deng, J., & Ye, Y. (2006). Stock enhancement and translocation of the shrimp *Penaeus chinensis* in China. *Fisheries Research, 80*(1), 67–79. https://doi.org/10.1016/j.fishres.2006.03.015

Wang, W., Wang, C., Sardans, J., Tong, C., Jia, R., Zeng, C., & Peñuelas, J. (2015). Flood regime affects soil stoichiometry and the distribution of the invasive plants in subtropical estuarine wetlands in China. *Catena, 128*, 144–154. https://doi.org/10.1016/j.catena.2015.01.017

Wang, X., Yi, T., Li, W., Xu, C., Wang, S., Wang, Y., Li, Y., & Liu, X. (2022). Anthropogenic habitat loss accelerates the range expansion of a global invader. *Diversity and Distributions, 28*(8), 1610–1619. https://doi.org/10.1111/ddi.13359

Wanless, R. M., Scott, S., Sauer, W. H. H., Andrew, T. G., Glass, J. P., Godfrey, B., Griffiths, C., & Yeld, E. (2010). Semi-submersible rigs: A vector transporting entire marine communities around the world. *Biological Invasions, 12*(8), 2573–2583. https://doi.org/10.1007/s10530-009-9666-2

Ward, N. L., & Masters, G. J. (2007). Linking climate change and species invasion: An illustration using insect herbivores. *Global Change Biology, 13*(8), 1605–1615. https://doi.org/10.1111/j.1365-2486.2007.01399.x

Warrant, E., Frost, B., Green, K., Mouritsen, H., Dreyer, D., Adden, A., Brauburger, K., & Heinze, S. (2016). The Australian Bogong moth *Agrotis infusa*: A long-distance nocturnal navigator. *Frontiers in Behavioral Neuroscience, 10*, 77. https://doi.org/10.3389/fnbeh.2016.00077

Warshafsky, Z. T., Tuckey, T. D., Vogelbein, W. K., Latour, R. J., & Wargo, A. R. (2019). Temporal, spatial, and biological variation of nematode epidemiology in American eels. *Canadian Journal of Fisheries and Aquatic Sciences, 76*(10), 1808–1818. https://doi.org/10.1139/cjfas-2018-0136

Wasowicz, P., Sennikov, A. N., Westergaard, K. B., Spellman, K., Carlson, M., Gillespie, L. J., Saarela, J. M., Seefeldt, S. S., Bennett, B., Bay, C., Ickert-Bond, S., & Väre, H. (2020). Non-native vascular flora of the Arctic: Taxonomic richness, distribution and pathways. *Ambio, 49*(3), 693–703. https://doi.org/10.1007/s13280-019-01296-6

Way, A. G. (2014). "A Cosmopolitan Weed of the World": Following Bermudagrass. *Agricultural History, 88*(3), 354–367. https://doi.org/10.3098/ah.2014.88.3.354

Weldon, C., du Preez, L. H., Hyatt, A. D., Muller, R., & Speare, R. (2004). Origin of the amphibian chytrid fungus. *Emerging Infectious Diseases, 10*(12), 2100–2105. https://doi.org/10.3201/eid1012.030804

Weslawski, J., & Kotwicki, L. (2018). Macro-plastic litter, a new vector for boreal species dispersal on Svalbard. *Polish Polar Research, 39*(1), 165–174. https://doi.org/10.24425/118743

Westphal, M. I., Browne, M., MacKinnon, K., & Noble, I. (2008). The link between international trade and the global distribution of invasive alien species. *Biological Invasions, 10*(4), 391–398. https://doi.org/10.1007/s10530-007-9138-5

Weyl, O. L. F., Ellender, B. R., Wassermann, R. J., Truter, M., Dalu, T., Zengeya, T. A., & Smit, N. J. (2020). Alien freshwater fauna in South Africa. In *Biological invasions in South Africa* (Vol. 14, pp. 151–182). Springer. https://doi.org/10.1007/978-3-030-32394-3_6

Wied, J. P., Perotto-Baldivieso, H. L., Conkey, A. A. T., Brennan, L. A., & Mata, J. M. (2020). Invasive grasses in South Texas rangelands: Historical perspectives and future directions. *Invasive Plant Science and Management, 13*(2), Article 2. https://doi.org/10.1017/inp.2020.11

Wilk-Woźniak, E., & Najberek, K. (2013). Towards clarifying the presence of alien algae in inland waters—Can we predict places of their occurrence? *Biologia, 68*(5), Article 5. https://doi.org/10.2478/s11756-013-0221-3

Williams, A. E., Duthie, H. C., & Hecky, R. E. (2005). Water hyacinth in Lake Victoria: Why did it vanish so quickly and will it return? *Aquatic Botany, 81*(4), 300–314. https://doi.org/10.1016/j.aquabot.2005.01.003

Williams, K., Ewel, K. C., Stumpf, R. P., Putz, F. E., & Workman, T. W. (1999). Sea-Level Rise and Coastal Forest Retreat on the West Coast of Florida, Usa. *Ecology, 80*(6), 2045–2063. https://doi.org/10.1890/0012-9658(1999)080[2045:SLRACF]2.0.CO;2

Williams, S. L., & Smith, J. E. (2007). A global review of the distribution, taxonomy, and impacts of introduced seaweeds. *Annual Review of Ecology, Evolution, and Systematics, 38*(1), 327–359. https://doi.org/10.1146/annurev.ecolsys.38.091206.095543

Willis, S. G., & Hulme, P. E. (2002). Does temperature limit the invasion of *Impatiens glandulifera* and *Heracleum mantegazzianum* in the UK? *Functional Ecology, 16*(4), 530–539. https://doi.org/10.1046/j.1365-2435.2002.00653.x

Wilson, H. E., Carroll, G. C., Roy, B. A., & Blaisdell, G. K. (2014). Tall fescue is a potential spillover reservoir host for Alternaria species. *Mycologia, 106*(1), Article 1. https://doi.org/10.3852/12-330

Wilson, S. B., & Mecca, L. K. (2003). Seed Production and Germination of Eight Cultivars and the Wild Type of *Ruellia tweediana*: A Potentially Invasive Ornamental. *Journal of Environmental Horticulture, 21*(3), 137–143. https://doi.org/10.24266/0738-2898-21.3.137

Wilson, S. D. (1988). The effects of military tank traffic on prairie: A management model. *Environmental Management, 12*(3), 397–403. https://doi.org/10.1007/BF01867529

Wittenberg, R., & Cock, M. J. W. (2003). *Invasive Alien Species: A Toolkit of Best Prevention and Management Practices.* CAB International. https://www.cabidigitallibrary.org/doi/10.1079/9780851995694.0000

Wolff, W. J., & Reise, K. (2002). Oyster Imports as a Vector for the Introduction of Alien Species into Northern and Western European Coastal Waters. In E. Leppäkoski, S. Gollasch, & S. Olenin (Eds.), *Invasive Aquatic Species of Europe. Distribution, Impacts and Management* (pp. 193–205). Springer Netherlands. https://doi.org/10.1007/978-94-015-9956-6_21

Woodruff, J. D., Irish, J. L., & Camargo, S. J. (2013). Coastal flooding by tropical cyclones and sea-level rise. *Nature, 504*(7478), Article 7478. https://doi.org/10.1038/nature12855

Work, T. T., McCullough, D. G., Cavey, J. F., & Komsa, R. (2005). Arrival rate of nonindigenous insect species into the United States through foreign trade. *Biological Invasions, 7*(2), 323–332. https://doi.org/10.1007/s10530-004-1663-x

World Bank, & International Monetary Fund. (2012). *Global Monitoring Report 2012: Food Prices, Nutrition, and the Millennium Development Goals.* The World Bank.

Wringe, B. F., Jeffery, N. W., Stanley, R. R. E., Hamilton, L. C., Anderson, E. C., Fleming, I. A., Grant, C., Dempson, J. B., Veinott, G., Duffy, S. J., & Bradbury, I. R. (2018). Extensive hybridization following a large escape of domesticated Atlantic salmon in the Northwest Atlantic. *Communications Biology, 1*(1), 1–9. https://doi.org/10.1038/s42003-018-0112-9

WTO. (2021). *World Trade Report* [Text]. https://www.wto-ilibrary.org/trade-monitoring/world-trade-report_0949bcf3-en

Wu, H., & Ding, J. (2019). Global Change Sharpens the Double-Edged Sword Effect of Aquatic Alien Plants in China and Beyond. *Frontiers in Plant Science, 10*, 787. https://doi.org/10.3389/fpls.2019.00787

Xavier, R. de O., Leite, M. B., & Silva-Matos, D. M. da. (2017). Stress responses of native and exotic grasses in a Neotropical savanna predict impacts of global change on invasion spread. *Austral Ecology, 42*(5), 562–576. https://doi.org/10.1111/aec.12475

Xi, K., Kumar, K., Holtz, M. D., Turkington, T. K., & Chapman, B. (2015). Understanding the development and management of stripe rust in central Alberta. *Canadian Journal of Plant Pathology, 37*(1), 21–39. https://doi.org/10.1080/07060661.2014.981215

Yager, L. Y., Jones, J., & Miller, D. L. (2009). Military Training and Road Effects on *Imperata cylindrica* (L.) Beauv. (Cogongrass). *Southeastern Naturalist, 8*(4), 695–708. https://www.jstor.org/stable/25599364

Yang, M., Lu, Z., Fan, Z., Liu, X., Hens, L., De Wulf, R., & Ou, X. (2018). Distribution of non-native plant species along elevation gradients in a protected area in the eastern Himalayas, China. *Alpine Botany, 128*(2), 169–178. https://doi.org/10.1007/s00035-018-0205-6

Zenni, R. D. (2014). Analysis of introduction history of invasive plants in Brazil reveals patterns of association between biogeographical origin and reason for introduction. *Austral Ecology, 39*(4), Article 4. https://doi.org/10.1111/aec.12097

Zenni, R. D., Dickie, I. A., Wingfield, M. J., Hirsch, H., Crous, C. J., Meyerson, L. A., Burgess, T. I., Zimmermann, T. G., Klock, M. M., Siemann, E., Erfmeier, A., Aragon, R., Montti, L., & Le Roux, J. J. (2017). Evolutionary dynamics of tree invasions: Complementing the unified framework for biological invasions. *AoB Plants, 9*(1). https://doi.org/10.1093/aobpla/plw085

Zenni, R. D., & Ziller, S. R. (2011). An overview of invasive plants in Brazil. *Revista Brasileira de Botânica, 34*(3), 431–446. https://doi.org/10.1590/S0100-84042011000300016

Zhang, G., Qu, H., Chen, G., Zhao, C., Zhang, F., Yang, H., Zhao, Z., & Ma, M. (2019). Giant discoveries of oil and gas fields in global deepwaters in the past 40 years and the prospect of exploration. *Journal of Natural Gas Geoscience, 4*(1), 1–28. https://doi.org/10.1016/j.jnggs.2019.03.002

Zhang, Q., Yang, R., Tang, J., & Chen, X. (2008). Competitive interaction between the invasive *Solidago canadensis* and native *Kummerowia striata* in lead contaminated soil. *Botanical Studies, 49*(4), 385–391.

Zhang, Z., Li, J., Li, Y., Liu, W., Chen, Y., Zhang, Y., & Li, Y. (2021). Spatially discontinuous relationships between salt marsh invasion and mangrove forest fragmentation. *Forest Ecology and Management, 499,* 119611. https://doi.org/10.1016/j.foreco.2021.119611

Zheng, Y. L., Burns, J. H., Liao, Z. Y., Li, Y. P., Yang, J., Chen, Y. J., Zhang, J. L., & Zheng, Y. G. (2018). Species composition, functional and phylogenetic distances correlate with success of invasive *Chromolaena odorata* in an experimental test. *Ecology Letters.* https://doi.org/10.1111/ele.13090

Ziska, L. H., Reeves, J. B., & Blank, B. (2005). The impact of recent increases in atmospheric CO_2 on biomass production and vegetative retention of Cheatgrass *(Bromus tectorum)*: Implications for fire disturbance. *Global Change Biology, 11*(8), 1325–1332. https://doi.org/10.1111/j.1365-2486.2005.00992.x

Zobel, D. B., Roroth, L., & Hawk, G. (1985). *Ecology, Pathology, and Management of Port-Orford-cedar (Chamaecyparis lawsoniana).* U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. https://doi.org/10.2737/PNW-GTR-184

Zozaya, S. M., Alford, R. A., & Schwarzkopf, L. (2015). Invasive house geckos are more willing to use artificial lights than are native geckos. *Austral Ecology, 40*(8), 982–987. https://doi.org/10.1111/aec.12287

Chapter 4

IMPACTS OF INVASIVE ALIEN SPECIES ON NATURE, NATURE'S CONTRIBUTIONS TO PEOPLE, AND GOOD QUALITY OF LIFE[1]

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IMPACTS OF INVASIVE ALIEN SPECIES ON NATURE, NATURE'S CONTRIBUTIONS TO PEOPLE, AND GOOD QUALITY OF LIFE

EXECUTIVE SUMMARY

1 Invasive alien species impact nature at all ecological levels, from native individuals, populations, species, to communities and ecosystems *(well established)* {4.3.1}. Although some invasive alien species can have both positive and negative impacts *(well established)* {4.3, 4.4, 4.5, 4.6}, the overall negative impacts of invasive alien species far exceed any positive impacts on nature and humans *(established but incomplete)* {4.3.1, 4.4.1, 4.5.1, 4.6.1, 4.6.3}. Almost three-quarters (71 per cent) of the documented impacts on nature adversely affect native species *(well established)* {4.3.1}. The magnitude of impacts of invasive alien species varies depending on the geographic and environmental context *(well established)* {4.3.1, 4.3.2, 4.3.3, 4.4.1, 4.4.2, 4.4.3, 4.5.1, 4.5.2, 4.5.3, 4.6.1, 4.6.2, 4.6.3, 4.6.4, 4.6.5}. The most commonly observed impacts on nature are changes in ecosystem properties, reductions in the performance of native species and declines in local populations of both plants and animals *(well established)* {4.3.1.3}. The most frequently observed mechanisms of impacts are competition, physical and chemical changes of the invaded ecosystems and trophic interactions through predation and herbivory *(well established)* {4.3.1.3}. In terrestrial ecosystems, most studies of impacts on nature are documented from plants and occur in forests, grasslands and human-dominated habitats *(well established)* {4.3.2.1}. Few impacts on nature are documented from very cold (tundra and high mountain habitats), very dry (deserts and xeric shrub lands) or flooded terrestrial habitats (wetlands – peatlands, mires, bogs) *(well established)* {4.3.1}. No impacts have been documented in the cryosphere and the deep-sea *(established but incomplete)* {4.3.1} **(Table 4.2)**. The magnitude of negative impacts of invasive alien species often varies with the invaded biomes and species, and impacts are sometimes exacerbated or attenuated by the interaction of invasive alien species with other drivers such as climate change, changes in land- and sea-use, or pollution *(established but incomplete)* {4.3.1} **(Box 4.5)**. The number of documented impacts of invasive alien species has risen in parallel with the documented number of alien species *(established but incomplete)* {4.3.1}. About 7 per cent of alien plants, 17 per cent of alien vertebrates, 23 per cent of alien invertebrates, and 12 per cent of alien microbes are known to be invasive, but their numbers are likely an underestimate *(established but incomplete)* {4.2}.

2 Invasive alien species have contributed to local or global extinctions of native species *(well established)* {4.3.1} (Box 4.4). Of all invasive alien species with documented impacts, 6 per cent (218 invasive alien species) have been associated with the local extinction of at least one native species *(established but incomplete)* {4.3.1}. Invasive alien species are a significant factor that directly or indirectly caused 60 per cent of documented global animal and plant extinctions *(established but incomplete)* **(Box 4.4)** and have caused 1,215 documented local extinctions of 255 native species across all taxa *(established but incomplete)* {4.3.1}. These local extinctions have been documented in marine (23.2 per cent) freshwater (14.5 per cent) and terrestrial realms (62.1 per cent) *(well established)* {4.3.1}. Invasive alien animals (vertebrates 51 per cent, invertebrates 32.5 per cent) are more often implicated in causing local extinctions than invasive alien plants (15.3 per cent) and microbes (1.2 per cent) *(well established)* {4.3.1}.

3 Impacts of invasive alien species are more harmful to isolated ecosystems, such as islands, than elsewhere *(established but incomplete)* {4.3.1.1}. Documented negative impacts on native species on islands are far more frequent than positive impacts (40.5 per cent vs. 4.5 per cent) *(well established)* {4.3.1.1}. Of the global extinctions caused by invasive alien species, the overwhelming majority occurred on islands and other isolated ecosystems *(established but incomplete)* {4.3.1} **(Box 4.4)**. Local extinctions are more frequently documented from islands than from non-island locations (9.2 per cent vs. 4.0 per cent) *(well established)* {4.3.1}. Of the top ten invasive alien species documented to have caused local extinctions on islands, five are domesticated or synanthropic species: *Rattus* spp. (rats), *Capra hircus* (goats), *Mus musculus* (house mouse), *Felis catus* (cat), but

also other vertebrates such as *Anas platyrhynchos* (mallard) *(well established)* {4.3.1.1}.

4 Invasive alien species pose a substantial threat to the conservation of native biodiversity, landscapes and seascapes in protected areas *(established but incomplete)* {4.3.1.2}. Invasive alien species impact areas protected for nature conservation, with impacts of similar magnitude and frequency occurring both inside and outside protected areas *(established but incomplete)* {4.3.1.2}. Impacts on nature in protected areas constitute 19.3 per cent of the total number of documented impacts on nature *(established but incomplete)* {4.3.1.2}. Reports of negative impacts on native species in protected areas are far more frequent than positive impacts (33.2 per cent vs. 6.3 per cent) *(established but incomplete)* {4.3.1.2}.

5 Invasive alien species cause impacts on all categories of nature's contributions to people *(well established)* {4.4}. A large majority (80 per cent) of documented impacts on nature's contribution to people are negative and harm people by decreasing ecosystem services *(well established)* {4.4.1}. The most commonly observed negative impact of invasive alien species to nature's contributions to people is a reduction of human food supply *(well established)* {4.4.1}, which is caused by all taxa, in all regions and realms *(well established)* {4.4.2, 4.4.3}. Other important impacts of invasive alien species on nature's contributions to people are on habitat maintenance (16 per cent records) and on the provision of materials, companionship and labour (14 per cent records). In terrestrial systems, the most common invasive alien species causing impacts are plants, particularly in cultivated areas and in temperate and boreal forests *(well established)* {4.4.2.1}. In inland waters, 70 per cent of the documented impacts on nature's contributions to people are from inland surface waters and water bodies/freshwater *(well established)* {4.4.2.2}, and most of them are caused by invasive alien vertebrates *(well established)* {4.4.2.2}. In marine systems, the impacts are mostly caused by invasive alien invertebrates and predominate in shelf ecosystems *(well established)* {4.4.2.3}.

6 Impacts of invasive alien species on human health vary from nuisance to poisoning, disease and death *(well established)* {4.5.1}. Zoonotic diseases transmitted by invasive mosquitos inflict misery, chronic disease and death *(well established)* {4.5.1.3}. Invasive alien plants can be highly allergenic or phytotoxic *(well established)* {4.5}. Several invasive ant species have been documented as causing serious allergic or toxic reactions *(well established)* {4.5.1.3}. Health impacts caused by venomous and poisonous invasive alien marine species have frequently been documented in the Mediterranean Sea *(well established)* {4.5.1.3}.

7 Global cumulative damages due to invasive alien species totalled more than US$ 1.738 trillion between 1970 and 2020 *(established but incomplete)* (Box 4.13). In 2017 alone, documented aggregate global costs of biological invasions were estimated to reach US$162.7 billion, exceeding the 2017 gross domestic product of 52 of the 54 countries on the African continent, and more than twenty times higher than the combined total funds available in 2017 for the World Health Organization and the United Nations *(established but incomplete)* **(Box 4.13)**. In 2019, global annual costs of biological invasions were estimated to exceed $423 billion, with variations across regions, but this is likely a gross underestimation *(established but incomplete)* **(Box 4.13)**. North America (53 per cent) and Asia (13 per cent) were associated with the highest documented costs, which is partly driven by cost data incompleteness for most taxa and regions of the world *(well established)* **(Box 4.13)**. Agriculture is the economic sector most frequently documented as affected by invasive alien species and specifically by insects which are often categorized as pests *(established but incomplete)* **(Box 4.13)**.

8 Invasive alien species cause impacts on good quality of life that affect the opportunities for people to live a fulfilled life *(established but incomplete)* {4.5}. The majority of the 3,783 documented impacts on good quality of life are documented as negative for people (about 85 per cent) *(established but incomplete)* {4.5.1}. Most negative impacts (56 per cent) on good quality of life are the result of changes to "material and immaterial assets" by invasive alien species *(established but incomplete)* {4.5.1, 4.5.2, 4.5.3}. Invertebrates are documented as causing the highest number of negative impacts on good quality of life (51 per cent of negative impacts) *(established but incomplete)* {4.5.3}. Conversely, plants (responsible for 42 per cent of positive impacts) are more likely to result in positive impacts on good quality of life *(established but incomplete)* {4.5.3}. Negative and positive impacts on society are most often documented in Asia-Pacific (41 per cent of negative impacts and 53 per cent of positive impacts), and in cultivated areas (29 per cent of negative impacts and 26 per cent of positive impacts) *(established but incomplete)* {4.5.2.1, 4.5.3}. Although there is very little systematic research on gender differences in impacts of invasive alien species, the available data suggest that some invasive alien species may cause gender-differentiated impacts *(established but incomplete)* {4.5.1}.

9 Indigenous Peoples and local communities report more negative than positive impacts caused by invasive alien species, especially on water resources, human health and health of livestock and access to traditional lands *(well established)* {4.6.1}. Indigenous Peoples and local communities report ten times more negative than positive impacts caused by invasive alien species on nature (92 per cent negative, 8 per cent positive) *(well established)* {4.6.1}. Impacts on nature, often affect the deep kinship connection that many Indigenous Peoples and local communities have with nature *(well established)* {4.6.3}. When considering nature's contributions to people, reports are more balanced (55 per cent negative to 45 per cent positive) *(well established)* {4.6.2}. Two-thirds (68 per cent) of the impacts on the good quality of life of Indigenous Peoples and local communities have been documented as negative, compared to one-third (32 per cent) that have been documented as positive *(well established)* {4.6.3}. Invasive alien species have frequently been documented to cause the loss of access to and mobility within traditional lands, leading to harder labour requirements *(well established)* {4.6.3}. Negative impacts on the health of Indigenous Peoples and local communities can be direct (e.g., injury) and indirect, including general feelings of despair and stress. Some invasive alien species can provide some benefits, including income and development of local industry *(well established)* {4.6.3, 4.6.4}, but Indigenous Peoples and local communities highlight that seemingly positive impacts are not often considered wholly positive by their communities, especially when communities had little agency or choice in responding to the invasive alien species *(well established)* {4.6.2, 4.6.3, 4.6.4}. There are many cases where Indigenous Peoples and local communities have adapted to the negative impacts of invasive alien species *(well established)* (4.6.3). Whilst more negative impacts have been documented on cultural values and practices, involvement of Indigenous Peoples and local communities in the use and management of invasive alien species is, in some cases, also documented as an opportunity for skills development and knowledge transfer *(established but incomplete)* {4.6.5}.

10 There are substantial geographic and taxonomic gaps in the documentation, quantification and understanding of impacts *(established but incomplete)* {4.7.2}. The quality and quantity of information available on impacts of invasive alien species for different taxa, units of analysis, regions and realms differ greatly, and research efforts are unevenly distributed across regions, temporal scales, and taxa *(well established)* {4.7.2}. These biases can be observed across all realms, especially in marine ecosystems, where the extent and timing of research efforts on marine invasive alien species lag behind terrestrial studies *(established but incomplete)* {4.7.2}. About 95 per cent of the sources listed in the dataset are in English, severely underrepresenting studies only available in non-anglophone sources *(well established)* {4.7.2}.

4.1 INTRODUCTION

> "The cardoon *(Cynara cardunculus)* has a far wider range: it now occurs in these latitudes on both sides of the Cordillera, across the continent. I saw it in unfrequented spots in Chile, Entre Rios, and Banda Oriental. In the latter country alone, very many (probably several hundred) square miles are covered with one mass of these prickly plants and are impenetrable by man or beast. Over the undulating plains, where these great beds occur, nothing else can live. Before their introduction, however, I apprehend the surface supported as in other parts a rank herbage. I doubt whether any case is on record, of an invasion of so grand a scale of one plant over the aborigines." (Darwin, 1839).

At the time Charles Darwin wrote this, European powers vied to import, grow and disseminate "exotic" plants and animals. The earliest "jardins d'acclimatation" were erected on the order of the King of France at the time, Louis XV, to accommodate edible, medicinal and decorative plants elsewhere; breadfruit from the South Pacific was shipped to French Guiana, and coffee plants to the Antilles and Brazil (Bailey, 2018). This was the continuation of a process rooted in prehistorical millennia. Zooarchaeological and archaeobotanical studies reveal the spread of the Near Eastern suite of domesticates, cultivated plants and synanthropic biota across Europe, Asia and Africa (Bortolus *et al.*, 2015; Colledge *et al.*, 2013; **Chapter 1**, **Figure 1.3**). Austronesian people transported their domesticated animals, including dogs, pigs, chickens and the synanthropic *Rattus exulans* (Pacific rat) to isolated archipelagos of Remote Oceania long before the sixteenth century (N. Amano *et al.*, 2021; Crabtree, 2016; Giovas, 2006). The direct and indirect impacts of these species on island ecosystems (**Glossary**), through agricultural deforestation and the introduction of mammalian predators, have only recently come to the fore: palaeoecological data reveal losses of many species (Drake & Hunt, 2009; Fillios *et al.*, 2012; Prebble & Wilmshurst, 2009).

Despite vast numbers of terrestrial, inland waters, and marine introductions over millennia, written documentation of their impacts was rare until the twentieth century. For example, *Sporobolus alterniflorus* (smooth cordgrass) occupying subtropical and temperate salt marshes along the Atlantic coast of South America may have been introduced in the eighteenth or early nineteenth century, but remained a "hidden invasion"; its impacts on coastal geomorphology and biodiversity have been overlooked and undocumented (Bortolus *et al.*, 2015). Studies that actually document impacts of invasive alien species have been limited to 3515 invasive alien species, about 10 per cent of all alien species (**Glossary**) according to the Global Register of Introduced and Invasive Species (GRIIS).

Charles Elton evinced great interest in biological invasions (**Glossary**) as early as the 1930s. Studying a bevy of introduced species in the United Kingdom, from *Ondatra zibethicus* (muskrat) to *Rattus norvegicus* (brown rat), he denounced them as a zoological catastrophe. Elton's seminal contribution (Elton, 1958) highlighted the impacts of invasive alien species – animals, plants, pathogens, terrestrial, aquatic and marine – and raised public awareness of biological invasions as a conservation issue (Simberloff, 2010). Still, it was not until the 1980s, when the Scientific Committee on Problems of the Environment (SCOPE) convened a series of workshops, that contemporary invasion biology was launched (Mooney & Drake, 1989; Simberloff *et al.*, 2013) and soon established that invasive alien species could have severe and lasting impacts on ecosystem functions, and that nearly every type of ecosystem had been affected (Lodge, 1993; **Chapter 1**, **Figure 1.2**). The mode, rate, order and crypticity of introduction, the inherent complexity in interactions between invasive alien species population and host community and ecosystem, and their interactions with the environment are each context-driven and difficult to assess (Jarić *et al.*, 2019; Parker *et al.*, 1999; Vanderhoeven *et al.*, 2017; **Chapter 1**, **section 1.5**).

Invasive alien species cause a wide array of economic damage, disrupting the production of goods and services. For example, invasive alien species can reduce timber and agricultural output (T. P. Holmes *et al.*, 2009; Paini *et al.*, 2016), damage infrastructure (Fritts, 2002), impact the operations of public utility companies (Elliott *et al.*, 2005; Magara *et al.*, 2001), and disrupt navigation (Ashe & Driscoll, 2013; Bryson *et al.*, 2008; Grewell *et al.*, 2016; Lindgren *et al.*, 2013; Mallison *et al.*, 2001). Invasive alien species are also notorious for altering nature's contributions to people and good quality of life (**Glossary**), which affects property values (Olden & Tamayo, 2014), tourism (Mejía & Brandt, 2015), and outdoor recreation (Lauber *et al.*, 2020). Human health can be profoundly affected, too (Juliano & Lounibos, 2005; Kemp *et al.*, 2000; World Health Organization & Convention on Biological Diversity, 2015). Significant costs are also associated with invasive alien species prevention and control efforts (**Glossary**), including clearing costs (Marais *et al.*, 2004) and increased costs of transportation (e.g., road-right-of-way maintenance costs, hull maintenance, inspection stations, ballast water treatment system costs, etc.). Invasive alien species are sometimes associated with economic benefits, having been deliberately introduced for aquaculture (De Silva *et al.*, 2009), forestry and landscaping (Knowler & Barbier, 2005; Richardson, 1998), cultural reasons (Pejchar & Mooney, 2009), or recreational pursuits such as sport fishing, yet there is general agreement that their net economic effect is overwhelmingly negative (Bradshaw *et al.*, 2016; Diagne, Leroy, *et al.*, 2021; Zenni *et al.*, 2021).

Though related literature has increased in recent years, research on the economic benefits and costs of invasive alien species is in its infancy. Perhaps more concerning, researchers are still trying to understand the links among biological invasions and economic activities, some of which are indirect and difficult to quantify (B. A. Jones & McDermott, 2018; B. A. Jones, 2016; Charles & Dukes, 2007). As a result, few long-term studies examine economic impacts over time (Essl *et al.*, 2011; Cuthbert, Pattison, *et al.*, 2021). As invasive alien species continue to spread and understanding about the economic implications of invasive alien species increases, it is safe to assume that the estimate of sustained damages will continue to rise. Perceptions of the costs and benefits of introduced species are varied (Jubase *et al.*, 2021; R. T. Shackleton, Richardson, *et al.*, 2019; Verbrugge *et al.*, 2013; **Chapter 1**, **section 1.5.2**). A limited number of invasive alien species are exploited commercially, though some of those have had substantial negative impacts in recipient ecosystems. In 1999, the International Union for the Conservation of Nature (IUCN), through its Invasive Species Specialist Group (ISSG), established the list of "100 of the world's worst invasive alien species" to increase public awareness (GISD, 2013). Geraldi *et al.* (2019), examined media attention to the aquatic and marine species on the IUCN list and concluded that coverage was low and short-lived; an important observation given the influence of media on societal environmental perceptions. Listed by the IUCN (but unexamined by Geraldi *et al.*, 2019) are *Oncorhynchus mykiss* (rainbow trout) and *Salmo trutta* (brown trout), which have been introduced worldwide for the main purpose of recreational fishing and have subsequently resulted in significant losses of biodiversity (Cambray, 2003). Yet, the growing popularity of sport fishing and angling delivers significant economic benefits to tourism, rending these and similarly introduced invasive fish "sacrosanct" amongst some stakeholders (J. E. Jackson *et al.*, 2004; Lewin *et al.*, 2006). Despite this inherent complexity, this chapter provides a global analysis and synthesis of the environmental, economic and social impacts of invasive alien species from available evidence (published peer-reviewed literature, grey literature, and information from Indigenous and local knowledge systems; **Glossary** and **Box 4.1**). This assessment only reflects documented impacts; however, the total impact of invasive alien species remains unknown.

The chapter flows from an unprecedented assessment of impacts, through which an impact database has been compiled. **Section 4.1** introduces the major concepts underpinning the analysis of impacts; **section 4.2** presents the methodology employed to record and analyse impacts in the chapter; **section 4.3** presents the analysis and synthesis of impacts on nature; **section 4.4** covers impacts on nature's contributions to people; and **section 4.5** describes impacts on good quality of life. For each section, the team of authors have presented general patterns, impacts by realm and units of analysis, impacts by region, and impacts by invasive alien species taxon. **Section 4.6** presents a

Box 4 1 **Rationale of the chapter.**

The chapter focuses on the impacts of invasive alien species on nature (**Glossary**) and nature's contributions to people and a good quality of life, as defined in the conceptual framework of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES; **Chapter 1**, **sections 1.6.1**), including non-economic values (e.g., cultural, social and shared, recreational, scientific, spiritual and aesthetic values).

Guiding questions:

- Which native taxa, nature's contributions to people and components of good quality of life are most negatively and positively impacted by invasive alien species?
- Which units of analysis and regions are most negatively and positively impacted by invasive alien species?
- Which invasive alien species caused local and global extinctions and which native species (**Glossary**) and taxonomic groups are affected?
- What are the global monetary costs of invasive alien species?
- How do people, including Indigenous Peoples and local communities, assess the magnitude of impacts of invasive alien species?
- What are our knowledge gaps and biases in the type and distribution of impacts across taxa, regions, units of analysis?

Keywords:
positive and negative impacts, invasive alien species, nature's contributions to people, good quality of life, native taxa, Indigenous Peoples and local communities, units of analysis.

summary of some impacts as perceived by Indigenous Peoples and local communities; and **section 4.7** discusses the future direction of impacts and their analysis, including the use of scenarios and modelling, and the knowledge gaps that can animate future improvements in methodology. Recording and analysing the impacts of invasive alien species will inform future efforts toward prevention and management (**Glossary**) of biological invasions.

4.1.1 Types of impacts: nature, nature's contributions to people, good quality of life

The impacts of invasive alien species on nature, nature's contributions to people, and good quality of life are all context-dependent, and range along a continuum from nearly indiscernible to region-wide changes (**Chapter 1**, **Figure 1.1** for definitions).

Impact on nature, formerly "ecological impact", is defined as a measurable change to the properties of an ecosystem (Ricciardi *et al.*, 2013), and implies that all introduced species can have an impact, even when not yet established or widespread (**Glossary**), which may vary in magnitude, simply by integration into the ecosystem. Impact can be measured at the level of an organism (e.g., effects on individual mortality and growth), a population (abundance), a community (species richness, evenness, composition, trophic structure), an ecosystem (physical habitat, nutrient cycling, contaminant cycling, energy flow), or a region (species richness, beta diversity). Individual, population and community-level impacts are most commonly studied (Jeschke *et al.*, 2014; Ricciardi *et al.*, 2013).

Impact on nature's contributions to people (**Chapter 1**, **Box 1.12**) comprises positive contributions as well as negative impacts, e.g., exacerbating fire hazards, soil erosion, allergenic pollen, zoonotic diseases, poisoning and envenomation (Vaz *et al.*, 2017). Regardless of taxon, ecosystem and region, invasive alien species alter nature's contributions to people by affecting populations, community dynamics, ecosystem processes, and abiotic variables. Yet, despite awareness of the susceptibility of nature's contributions to people to alteration by invasive alien species, research has lagged behind and impacts are often overlooked or underappreciated, leaving threats to people unquantified (Charles & Dukes, 2007).

Each constituent of good quality of life (**Chapter 1**, **Table 1.4**) is vulnerable to alteration by invasive alien species. Changes to the constituents of good quality of life such as material and immaterial assets (e.g., the provisioning of food and fuel), safety, health, economic and cultural practices, social relations, or freedom of choice and action can affect peoples' lives (**Box 4.9** in **section 4.3.2.1**, for example).

Appreciation of the extent and intensity of impacts is essential for prioritizing appropriate policy and governance (**Glossary**) responses to invasions. Attention of policymakers, stakeholders and the public is focused on a subset of introductions perceived as "harmful", having resulted in extinction or extirpation of native species and/or striking changes to ecosystem functioning, nature's contributions to people and good quality of life (Simberloff *et al.*, 2013).

4.1.2 Directionality of impacts: nature, nature's contributions to people, good quality of life

Impact directionality (i.e., whether impacts of invasive alien species are assessed as "negative" or "positive") is partly grounded in subjective perceptions embedded within economic, cultural and social contexts. Perception of impacts as positive or negative depends on value systems and values can vary even within the same economic, cultural and social context (R. T. Shackleton, Richardson, *et al.*, 2019). Thus, there are different ways of defining whether nature or its elements are harmed or benefit.

Nature and its elements have intrinsic value, and one could argue that this value can be damaged by invasive alien species. Extinctions and extirpations caused by the unintentional introductions of rats, snakes, gypsy moths, and chestnut blight, can be considered negative impacts (Czech & Krausman, 1999; Butchart *et al.*, 2006; Kochalski *et al.*, 2019). Local population losses and niche contraction of native species may not induce immediate extirpation, but they augur reduction of genetic diversity, loss of functions, processes, and habitat structure, increasing the risk of decline and extinction (**Glossary**; Galil, 2007). When studies document cases of rising species richness or abundance of native species following introductions of an invasive alien species (Irigoyen *et al.*, 2011; McQuaid & Griffiths, 2014; Thomsen, 2010), they can be considered as positive impacts on nature. However, this assessment recognises that the purported benefits can be predicated on provision of novel habitat (e.g., polychaete and oyster reefs in muddy habitats, algal meadows) by transforming entire habitats to the detriment of the pre-existing community. In many communities, some native species suffer from the introduction of invasive alien species while others may benefit.

In this assessment, impacts on nature are defined as negative when a native species suffers disadvantage,

Box 4 2 Environmental and Socio-Economic Impact Classification for Alien Taxa: EICAT and SEICAT.

The International Union for Conservation of Nature (IUCN) EICAT framework was developed to categorize and assess negative impacts caused by alien taxa on native taxa (IUCN, 2020). The framework assesses how much a native species is affected by an invasive alien species. Other types of environmental impacts such as changes caused by alien taxa to abiotic ecosystem properties (e.g., soil or water chemistry) are considered under the framework only if such changes lead to a decrease in attributes of native biodiversity.

The EICAT classifies impacts in a 5-step semi-quantitative scale based on the level of biological organization affected (individuals → populations → communities), and the magnitude and reversibility of these impacts (Blackburn *et al.*, 2014). The five steps reflect an increase in the order of magnitude of the particular impact so that a new level of biological organization is involved. **Minimal Concern** – negligible impacts, and no reduction in performance of a native taxon's individuals; **Minor** – performance of individuals reduced, but no decrease in population size; **Moderate** – native taxon population decline; **Major** – native taxon local extinction (i.e., change in community structure), which is naturally reversible; and **Massive** – naturally irreversible local or global extinction of a native taxon (**Figure 4.1**; IUCN, 2020; Volery *et al.*, 2020). Impacts of invasive alien species can be caused through 10 mechanisms **(Figure 4.1)**. The EICAT is conceptually and structurally related to the IUCN Red List of Threatened Species, with the Red List categorizing a focal native species based on its risk of extinction, and the EICAT categorizing a focal alien taxon based on the degree to which it has negatively impacted native taxa (Van der Colff *et al.*, 2020).

The SEICAT assesses negative impacts of invasive alien species on good quality of life (Bacher *et al.*, 2018). It follows an approach similar to the EICAT. In particular, it classifies changes in human activities caused by invasive alien species into one of 5 magnitudes. These are: **Minimal Concern** – negligible impacts, and no reduction in individual peoples' activities; **Minor** – normal activities are more difficult, but no decrease in activity size, i.e., all people still carry out the activity; **Moderate** – decline in activity size, i.e., fewer people participate in an activity; **Major** – local disappearance of an activity from all or part of the area invaded by the invasive alien species, which is naturally reversible; and **Massive** – local irreversible disappearance of an activity from all or part of the area invaded by the invasive alien species (Bacher *et al.*, 2018). Changes in human activities can be caused through impacts on five constituents of good quality of life **(Box 4.3)**. The framework is based on the capability approach of welfare economics (Robeyns, 2005; Sen, 1999) and thus avoids ambiguities in interpreting impacts based on monetary approaches (Hoagland & Jin, 2006).

The EICAT and the SEICAT have been used to compare impact magnitudes of alien taxa at various spatial scales, across geographic regions and taxonomic groups (e.g., Evans *et al.*, 2016, 2020; Canavan *et al.*, 2019; Galanidi *et al.*, 2018; Kesner & Kumschick, 2018; Volery *et al.*, 2021), and to facilitate evidence-based prioritization and other management decisions (Rockwell-Postel *et al.*, 2020). Widespread application of both schemes is expected to reduce data biases and data gaps on the impacts of invasive alien species on nature and good quality of life. Recently, the EICAT framework was expanded to include a classification for positive impacts of invasive alien species for nature (EICAT+; Vimercati *et al.*, 2022) but this was not available at the time when data for this chapter were gathered. EICAT+ might allow comparison of positive and negative environmental impacts in a common framework for a better understanding of the consequences of invasive alien species and to better inform conservation decisions. For a comprehensive understanding and efficient management, the reporting of both negative and positive impacts is critical (Vimercati *et al.*, 2020).

Box 4 2

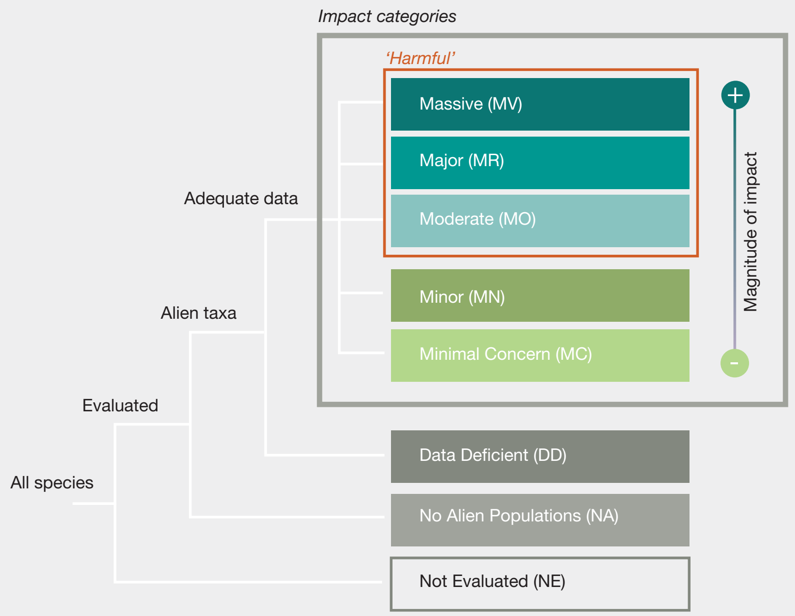


Figure 4 1 **EICAT and SEICAT categories and the relationship between them.**

The five impact categories (from minimal concern to massive) can be used to assess negative impacts caused by invasive alien species (which are a subset of alien species). Source: IUCN (2020), https://doi.org/10.2305/IUCN.CH.2020.05.en, under license CC BY-NC 4.0.

following the Environmental Impact Classification for Alien Taxa (EICAT, **Box 4.2**) approach developed by IUCN (2020), and as positive when a native species benefits from ecosystem changes due to the introduction of an invasive alien species (Vimercati *et al.*, 2022). However, not every ecosystem change can be assigned a unique directionality. For example, abiotic characteristics of ecosystems (e.g., changes in soil or water chemistry, structural complexity) can increase or decrease due to the impacts of an invasive alien species, but it is not straightforward to assign an impact direction (positive or negative) because these changes might have different consequences for different species. Moreover, abiotic ecosystem changes can sometimes be quantified as either an increase or a decrease of an indicator (e.g., an increase in the concentration of hydrogen ions (H^+) is equivalent to a decrease of the pH). Thus, in this report, impacts describing abiotic changes in ecosystem characteristics are classified as negative or positive only if the consequence of these changes is documented to harm or benefit a native species. Abiotic ecosystem changes are not assigned a directionality when it is unknown if a native species suffers or benefits from these changes. Invasive alien species can benefit or harm people, which determines the directionality of impacts on nature's contributions to people and good quality of life. Directional changes in nature's contributions to people (i.e., increases or decreases of the parameters that are measured, **Chapter 1**, **Box 1.12**) may be positively or negatively associated with changes in good quality of life. For this report, benefits in nature's contributions to people are documented as an increase in services and/or decrease in disservices, whereas deleterious changes would do the opposite (Vaz *et al.*, 2017). By contrast, in this report, directionality in good quality of life is assessed by changes in constituents of good quality of life (Bacher *et al.*, 2018; **Chapter 1**, **Table 1.4**) which are directly associated with humans profiting or suffering from the impacts of an invasive alien species. This follows the Socio-Economic Impact Classification for Alien Taxa (SEICAT; **Box 4.2**) approach, which recognizes that different people may perceive impacts by invasive alien species in different ways (Bacher *et al.*, 2018). Invasive alien species may cause a range of impacts with different directionality ("negative" and "positive") on native species of the resident community, on categories of nature's contributions to people, and on components of good quality of life. For example, a negative impact of an invasive alien species on a native predator may profit its native prey; an invasive alien species may increase food production at the expense of soil deterioration; or an invasive alien rangeland plant like *Echium plantagineum* (Paterson's curse) may profit bee keepers due to its proliferous nectar production, but be toxic to livestock and thus detrimental to farmers (Harris, 1984). Thus, impacts on nature can be at odds with impacts on nature's contributions to people and good quality of life. For instance, *Gambusia affinis* (western mosquitofish) has been widely introduced as a biological control agent (**Glossary**) to manage mosquito populations but also preys on rare indigenous fish, amphibians and invertebrates (Englund, 1999; Leyse *et al.*, 2004; Segev *et al.*, 2009; Rupp, 1996).

Occasionally, invasive alien species impact all three **(Table 4.1)** – nature, nature's contributions to people and good quality of life – as is the case with *Acacia mearnsii* (black wattle), in South Africa. This highly invasive alien species manifests significant negative impacts on water resources (losses estimated at 577 million m^3 annually), biodiversity, and the stability and integrity of riparian ecosystems, while supplying an industry of tanning agents and providing rural communities with firewood and building materials. Plantation owners, small growers and rural communities benefit economically from the products of invasive alien wattles, whereas most sectors of society bear the social and monetary costs of loss in water and biodiversity, and increase in fire risk and erosion (de Wit *et al.*, 2001).

When interpreting invasive alien species impacts, care should be taken to examine them in a comprehensive manner, addressing nature, nature's contributions to people, good quality of life, and their directionality. Reporting all types of impacts, positive and negative, separately allows a comprehensive picture of impacts by invasive alien species and avoids some impacts being masked by tallying or calculating "net impacts". For example, economic benefits are often gained by a few people or sectors while costs, often long-term ones, are borne by many others (Gozlan & Newton, 2009; Kelsch *et al.*, 2020).

The IPBES invasive alien species assessment acknowledges that the outcomes of assessments of the positive impacts of invasive alien species do not balance or offset their negative impacts, which may be irreversible (Lockwood *et al.*, 2023). Positive and negative stacked bar charts in this chapter do not imply that positive and negative impacts can be summed.

4.1.3 Impacts and Indigenous and local knowledge

Some Indigenous Peoples and local communities, because of their holistic and interconnected relationships with nature (M. C. C. Holmes & Jampijinpa, 2013) and close dependence on nature for livelihoods and support systems (Mungatana & Ahimbisibwe, 2012), experience impacts of invasive alien species that go beyond changes to distinct species or habitats, to include both negative and positive economic, social and cultural impacts, including on good quality of life (Vaarzon-Morel, 2010; Sundaram *et al.*, 2012; Jevon & Shackleton, 2015; K. Smith *et al.*, 2010; dos Santos *et al.*, 2014; Atyosi *et al.*, 2019; Martínez & Manzano-García, 2019; R. T. Shackleton, Shackleton, *et al.*, 2019). For some Indigenous Peoples and local communities, an impact of invasive alien species may change material assets, such as food and materials to sustain livelihoods (K. Smith *et al.*, 2010), as well as some immaterial values, including cultural practices (Monterroso *et al.*, 2011), opportunity for learning and teaching on traditional lands (Bach *et al.*, 2019), and persisting spiritual identities (Fischer, 2007), all of which underpin their health and well-being (Sangha *et al.*, 2015).

Reviews, such as the one conducted by Pfeiffer and Voeks (2008), highlight the importance of time scale in assessing impacts, and estimate that where an invasive alien species has been present for at least 3 generations (100 years plus), Indigenous Peoples and local communities may incorporate the invasive alien species into rituals, practices or as a resource. However, other studies (R. T. Shackleton *et al.*, 2017) and frameworks (C. M. Shackleton *et al.*, 2007) focused on sustainable livelihoods more broadly, suggest that if an invasive alien species is left unchecked over time, the negative impact on livelihoods and vulnerability of communities increases with longer exposure to the invasive alien species. For example, the Botswana San once embraced *Prosopis juliflora* (mesquite), planted by forestry officials in the 1980s, as a useful resource but, by the 1990s, its spread threatened animals, water sources and movement through the bush. Thus, the San people have since worked actively to eradicate (**Glossary**) it (Bach *et al.*, 2019; Fischer, 2007; Monterroso *et al.*, 2011; Mosweu *et al.*, 2013; Sangha *et al.*, 2015; K. Smith *et al.*, 2010).

The nature of research on impacts of invasive alien species for Indigenous Peoples and local communities has also developed over time. Early studies often documented the knowledge and use of invasive alien species by Indigenous Peoples and local communities (i.e., ethnobotanical studies looking at medicinal or food use of alien plants; Bye, 1981), while more complex impacts were not documented. As Indigenous and local knowledge has been elevated within mainstream arenas to inform global biodiversity policies (e.g., Local Biodiversity Outlooks; Forest Peoples Programme *et al.*, 2016, 2020), studies of Indigenous and local knowledge on broad ecological, social and cultural impacts have increased, including co-designed studies with Indigenous Peoples and local communities (e.g., Sloane *et al.*, 2021; S. Russell *et al.*, 2020). However, this recent rise in the number and complexity of studies does not mean that impacts have only recently been felt. For invasive alien species that arrived and spread centuries ago, the information about first impacts may not have been passed down through the generations of Indigenous Peoples or local communities, particularly if the introduced species is not part of ancestral or "Dreaming" stories and customs (Crowley, 2014; Salmón, 2000). Therefore, cultural stories and knowledge transferred in modern times may be more on how to use and adapt to invasive alien species, rather than documented negative impacts (e.g., rabbits in Australia, feral pigs in Hawaii, wild horses in North America, water hyacinth in waterways in Asia and Africa; Pfeiffer & Voeks, 2008; Collin, 2017).

Given the complexity of impacts considering time-scale and the diversity of Indigenous Peoples and local communities and their livelihoods, Pfeiffer and Voeks (2008) proposed a framework of invasive alien species impacts as either "impoverishing, augmenting, or facilitating" culture. Fitting within this framework, some studies recognize negative impacts of invasive alien species to the livelihoods of Indigenous Peoples and local communities (Kent & Dorward, 2015; Ngorima & Shackleton, 2019), others acknowledge the positive impacts such as facilitation of inter-generational culture retention (Maldonado Andrade, 2019), and some studies highlight the adaptation of some Indigenous Peoples and local communities to invasive alien species (P. L. Howard, 2019).

In this chapter, Indigenous and local knowledge sources have been included as data in the main impacts database (**section 4.2**) and, in addition, **section 4.6** includes a supplementary review of impacts directly documented by Indigenous Peoples and local communities from 124 peer-reviewed sources, which fills some information gaps in the mainstream database methods (**section 4.6**).[2]

4.2 METHODOLOGY

Authors of this chapter have systematically reviewed relevant available information to understand the impact of invasive alien species on nature, nature's contributions to people and good quality of life, at a global level for a large number of organisms and habitats.

Some regions of the world have notably more information in scientific publications than others (Nuñez *et al.*, 2019; Nuñez & Amano, 2021). Therefore, methods for reviewing literature varied within this chapter, with tailored criteria and systematic approaches for literature searches being adopted for different regions and taxa. The specific methodologies are presented in more detail in the data management report.[3] Reviewed information included scientific literature (papers, books) and grey literature (institutional reports, reports of agencies and other relevant sources), including from Indigenous and local knowledge, and databases of invasive alien species (e.g., Centre for Agriculture and Bioscience International (CABI)'s Invasive Species Compendium and the Global Invasive Species Database (GISD), the IUCN Red List of Threatened Species, or the InvaCost database).

From each analysed document, gathered data included:

- The geographical location of the impact.
- The corresponding IPBES unit of analysis (**Chapter 1**, **section 1.6.5** for a description of all 17 units of analysis), recording whether the impacted area was on an island or in a protected area.
- The name of the invasive alien species, and the name (if possible) and taxonomic group (plant, invertebrate, vertebrate, and microbe) of native species affected, as described in the document, and if the species was intensively used for multiple purposes by humans.
- The mechanism, magnitude (only for negative impacts) and direction of impacts on nature (**section 4.1.2**), at the local population level.
- The mechanism and direction of impacts on nature's contributions to people (**section 4.1.2**, **Box 4.3** and **Chapter 1**, **Box 1.12**). This doesn't include the magnitude of impacts on nature's contributions to people, as no standard methodology has been developed to date to assess it.
- The direction, magnitude (only for negative impacts) of impacts and affected constituents of good quality of life (**section 4.1.2**).
- The relation to Indigenous and local knowledge.

Authors did not collect data on the synergistic effects of other drivers of change in nature such as climate change (**Box 4.5**, in **section 4.3.1**), evolutionary aspects (**Box 4.8**, in **section 4.3.1.4**) or information on the interactions with other native or alien species (**Box 4.5**, in **section 4.3.1**).

The database of impacts developed through this chapter contains data on 24,129 reports of impacts caused by 3,515 invasive alien species, representing 10.9 per cent of all alien species (ranging from 5.5 per cent to 22.4 per cent, depending on the taxonomic groups, **Table 4.1**). There were no studies of impacts for many alien species and the real percentages of invasive alien species causing impacts is likely to be higher than documented in this chapter.
All numbers presented in this chapter are based on this single database compiled specifically for this chapter if not stated otherwise.

2. Data management report available at: https://doi.org/10.5281/zenodo.5760266
3. Data management report available at: https://doi.org/10.5281/zenodo.5766069

Box 4 3 Important terms and concepts used in this chapter.

Constituents of good quality of life are material and immaterial assets; safety; health; social and cultural relationships; freedom of choice and action (**Chapter 1**, **Table 1.4**).

Impacts on ecosystem properties are changes to (abiotic) ecosystem parameters e.g., soil variables, while it remains unknown how native species are affected by these changes (**section 4.1.1**).

Impacts on good quality of life (**Chapter 1**, **section 1.6.7.2**) can be positive or negative through changes in constituents of good quality of life; measured as changes in peoples' activities following the SEICAT approach (Bacher *et al.*, 2018).

Impact magnitudes:

- **Impact magnitudes on nature** follow the EICAT system from the IUCN (2020) namely impacts on performance of native individuals, population declines, local or global extinctions.
- **Impact magnitudes on good quality of life** are classified according to the SEICAT approach (Bacher *et al.*, 2018) namely human activities are more difficult, some people stop certain activities, and activity is locally abandoned.

Mechanisms include negative impacts on native species and follow the EICAT system from the IUCN (2020):

- **Competition** – the alien taxon competes with native taxa for resources (e.g., food, water, and space), leading to deleterious impact on native taxa.
- **Predation** – the alien taxon predates on native taxa, leading to deleterious impact on native taxa.
- **Hybridization** – the alien taxon hybridizes with native taxa, leading to deleterious impact on native taxa.
- **Transmission of disease** – the alien taxon transmits diseases (alien or native) to native taxa, leading to deleterious impact on native taxa.
- **Parasitism** – the alien taxon parasitizes native taxa, leading to deleterious impact on native taxa.
- **Poisoning/toxicity** – the alien taxon is toxic, or allergenic by ingestion, inhalation or contact, or allelopathic to plants, leading to deleterious impact on native taxa.
- **Bio-fouling** or other direct physical disturbance – the accumulation of individuals of the alien taxon on the surface of a native taxon (i.e., biofouling), or other direct physical disturbances not involved in a trophic interaction (e.g., trampling, rubbing, etc.) leads to deleterious impact on native taxa.
- **Grazing/herbivory/browsing** – grazing, herbivory or browsing by the alien taxon leads to deleterious impact on native taxa.
- **Chemical, physical, structural impact on ecosystem** – the alien taxon causes changes to the chemical characteristics of the native environment (e.g., pH; nutrient and/or water cycling), the physical characteristics of the native environment (e.g., disturbance or light regimes), or changes to the habitat structure (e.g., changes in architecture or complexity), leading to deleterious impact on native taxa.
- **Indirect impacts through interactions with other species** – the alien taxon interacts with other native or alien taxa (e.g., through any mechanism, including pollination, seed dispersal, apparent competition, mesopredator release), facilitating indirect deleterious impact on native taxa.

Nature's contributions to people are composed of 18 categories (**Chapter 1**, **Box 1.12**; Díaz *et al.*, 2018). Note that changes in nature's contributions to people do not always directly translate into positive or negative changes for people (e.g., if people do not use the increase in nature's contributions to people, then there is no actual contribution). Nature's contributions to people impacts are documented as positive or negative without assignment of magnitude, i.e., positive means an increase in nature's contributions to people, negative a decrease.

Positive impacts are assigned as positive when an entity profits from the change, i.e., a native species (impacts on nature) or humans (nature's contributions to people, good quality of life impacts).

Unit of analysis have been adopted by this assessment to classify "habitats" (**Chapter 1**, **section 1.6.5**).

Of the 3,515 invasive alien species that were found to cause impacts, 1,673 (48 per cent) cause impacts on nature, 1,530 (44 per cent) on nature's contributions to people, and 1,032 (29 per cent) on good quality of life. Invasive alien species frequently cause more than one type of impact: 556 invasive alien species (16 per cent) cause impacts on both nature and nature's contributions to people, 235 species (7 per cent) on both nature and good quality of life. About 5 per cent of all invasive alien species cause impacts on all three categories.

There are similar numbers of impacts documented from the Americas (8,163 reports), Europe and Central Asia (7,481), and Asia-Pacific (6,016), but considerably fewer reports from Africa (1,725) **(Figure 4.2)**. Of all the documented impacts, 4,679 are from islands, and 3,324 from protected areas. Most documented impacts are from the terrestrial realm (18,011, 74.6 per cent) with considerably fewer from aquatic realms (inland waters: 3,299, 13.7 per cent; marine: 2,352, 9.7 percent); 467 of the documented impacts were from studies that did not specify the realm **(Figure 4.2)**. Invasive alien species have been documented to cause impacts across all units of analysis, but most reports are from temperate and boreal forests and woodlands, inland waters and cultivated areas (including cropping, intensive livestock farming; **Table 4.2**). There are very few

Table 4 1 **Number of established alien species and invasive alien species identified in this assessment by taxonomic group.**

Data sources for the numbers of established alien species from different taxonomic groups: **Chapter 2**, **Table 2.3**). A subset of established alien species are known to cause adverse impacts; they are termed invasive alien species. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

	Taxonomic group				
	Plants	**Invertebrates**	**Vertebrates**	**Microbes**	**All taxa**
Number of established alien species	19,365	8,282	3,242	1,257	32,146
Number of established alien species with documented impacts	1,061	1,852	461	141	3,515
Percentage of invasive alien species	5.5%	22.4%	14.2%	11.2%	**10.9%**

Table 4 2 **Number of documented impacts across IPBES units of analysis.**

A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

IPBES units of analysis	Number of impact records
Tropical and subtropical dry and humid forests	2,664
Temperate and boreal forests and woodlands	3,849
Mediterranean forests, woodlands and scrub	1,248
Tundra and high mountain habitats	205
Tropical and subtropical savannas and grasslands	1,106
Temperate grasslands	2,147
Deserts and xeric shrublands	579
Urban/semi-urban	1,480
Cultivated areas (incl. cropping, intensive livestock farming etc.)	3,032
Aquaculture areas	144
Wetlands – peatlands, mires, bogs	728
Inland surface waters and water bodies/freshwater	3,107
Shelf ecosystems (neritic and intertidal/littoral zone)	2,295
Open ocean pelagic systems (euphotic zone)	7
Coastal areas intensively used for multiple purposes by humans	649
Cryosphere	-
Deep-sea	-

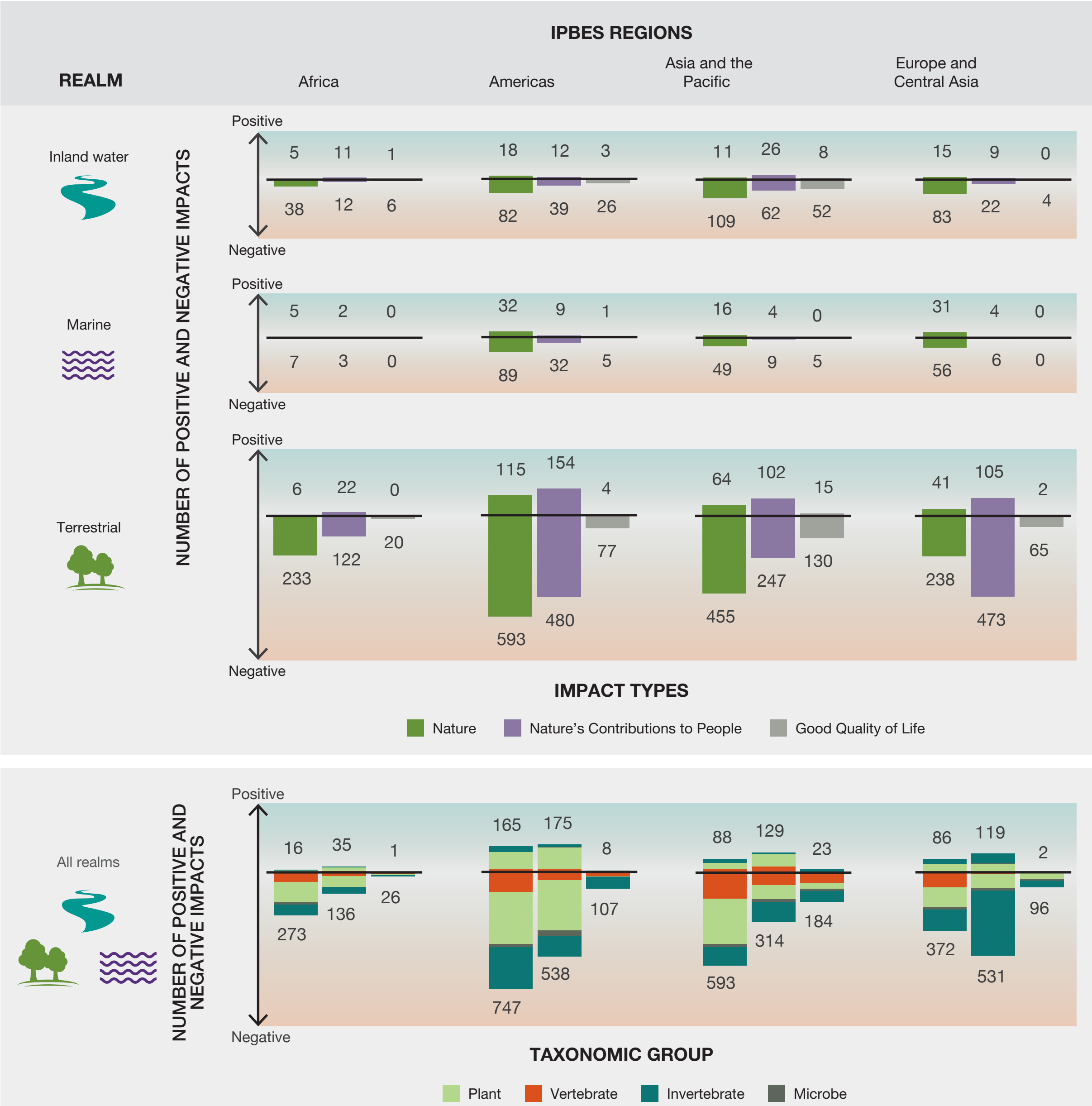


Figure 4 2 **Number of invasive alien species with documented negative and positive impacts on nature, nature's contribution to people and good quality of life, by taxonomic group, realm and IPBES region.**

Numbers and bars above indicate invasive alien species with positive impacts, numbers and bars below the x axes indicate species with negative impacts. Positive and negative stacked bar charts do not imply that positive and negative impacts can be summed. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

documented impacts from the open ocean, and no reports from the deep sea and the cryosphere were found. Twenty per cent of all impacts are reported from islands.

The most frequently documented impacts of invasive alien species are caused by plants (10,091 documented impacts), followed by invertebrates (8,180 documented impacts) and vertebrates (5,182 documented impacts), with invasive alien microbes having the lowest number of impact reports (676 documented impacts) **(Figure 4.2)**. These include all types of impacts (nature, nature's contributions to people, good quality of life) and all directions (positive, negative, and those that cannot be assigned a direction). Of the 13,898 documented impacts of invasive alien species on ecosystems properties, most affected native plants (6,376 documented impacts), followed by native invertebrates (4,629 documented impacts) and vertebrates (3,576 documented impacts), with impacts on native microbes being considerably less often documented (312).

4.3 IMPACTS OF INVASIVE ALIEN SPECIES ON NATURE

4.3.1 General patterns

Invasive alien species impact nature globally, and the majority of documented impacts are negative. This chapter documents more than 15,000 impacts on nature caused by 3,515 invasive alien species (**section 4.2**). Only a subset of these can be classified with a direction as being either negative, neutral or positive for native species (**section 4.1.2**). Of all the documented impacts with an assigned direction, 85 per cent (10,822) can be considered as negative impacts (caused by 1,623 species), and only 15 per cent (1,976) can be considered as positive impacts (caused by 361 species). Impact on ecosystem properties caused by 1,560 invasive alien species cannot be classified as either positive or negative impacts.

The vast majority of impacts were documented after the year 2000 **(Figure 4.3)**, which is likely to be a consequence of an increase in impacts correlated with the increase in number and occurrence of invasive alien species globally (**Chapter 2**, **section 2.2.1**), but is also due to an increase in research on the impact of invasive alien species. Negative impacts on nature have been documented since the beginning of the twentieth century with an almost exponential increase through time, while positive impacts of invasive alien species only started being documented in the 1970s **(Figure 4.3)**.

Invasive alien species most often documented causing impacts on nature

Invasive alien species with most records of negative impacts on nature include many vertebrates, e.g., terrestrial mammals such as *Rattus Rattus* (black rat), *Rattus exulans* (Pacific rat), *Felis catus* (cat), and *Vulpes vulpes* (red fox); *Rhinella marina* (cane toad); or marine and inland waters fishes such as *Pterois volitans* (red lionfish) and *Cyprinus carpio* (common carp). The ten most-often documented invasive alien species with negative impacts on nature include several ant species such as *Solenopsis invicta* (red imported fire ant), *Linepithema humile* (Argentine ant), and *Anoplolepis gracilipes* (yellow crazy ant); and *Procambarus clarkii* (red swamp crayfish). Examples of plants with many documented negative impacts on nature include *Reynoutria japonica* (Japanese knotweed) and *Lantana camara* (lantana).

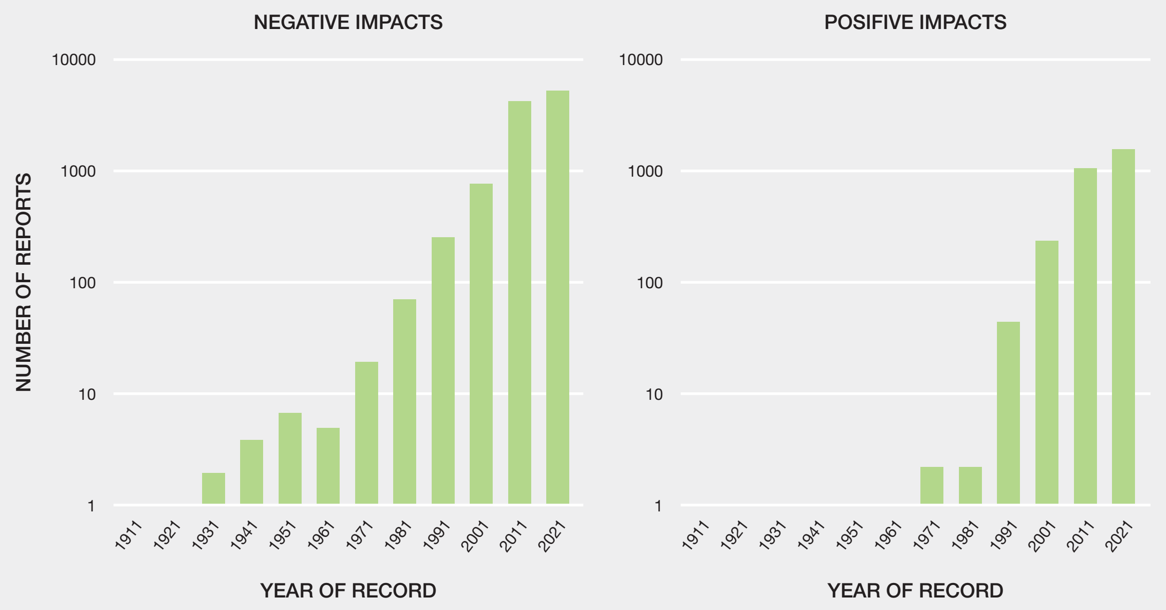


Figure 4 3 **Number of reported negative and positive impacts (y axes) on nature over time (x axis), from published literature since 1900 (note the logarithmic scale).**

A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Most positive impacts on nature are caused by plants and invertebrates. Terrestrial plants, such as *Solidago gigantea* (giant goldenrod), *Reynoutria japonica* (Japanese knotweed), or *Carpobrotus* spp. (iceplant), and trees like *Acacia longifolia* (golden wattle) are abundant nectar sources for many native insect species. Marine species such as *Caulerpa cylindracea* (green algae) provide habitat for native species. Aquatic invertebrates such as *Magallana gigas* (Pacific oyster), *Dreissena* spp. (zebra and quagga mussels), *Ficopomatus enigmaticus* (tubeworm), or *Didemnum vexillum* (carpet seas quirt) also impact positively nature by creating habitat, and sometimes providing food, for native species. Although there are no vertebrates among the top ten invasive alien species with most records of positive impacts on nature, those with the most frequent documented positive impacts are *Neogobius melanostomus* (round goby), which provides food for native species (Hempel *et al.*, 2016; Rakauskas *et al.*, 2020), and *Rhinella marina* (cane toad), which indirectly favors native medium-sized predators by reducing populations of their competitors and/or top predators (Brown *et al.*, 2011; Doody *et al.*, 2013).

A total of 280 (8 per cent) invasive alien species have been documented to cause both negative and positive impacts on nature (**section 4.1.2**). Among these are many of the species that most often have been documented causing negative impacts, such as *Dreissena* spp. (zebra and quagga mussels), *Reynoutria japonica* (Japanese knotweed), or *Rhinella marina* (cane toad).

Local extinctions

Some invasive alien species cause local extinctions of native populations. Six per cent (218 species) of all invasive alien species with documented impacts have caused a total of 1,215 local extinctions of native populations. Local

Table 4 3 **Number of local extinctions caused by invasive alien species by taxonomic group and realm.**

Number of documented local extinctions caused by invasive alien species for invertebrates, microbes, plants and vertebrates in different realms. Two local extinctions (plant, microbe) could not be assigned to a realm. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

	Number of local extinctions in the marine realm	Number of local extinctions in the terrestrial realm	Number of local extinctions in the inland waters realm	Number of local extinctions in all realms
Local extinctions caused by invasive alien plants	98	53	35	187
Local extinctions caused by invasive alien invertebrates	134	210	50	394
Local extinctions caused by invasive alien vertebrates	48	480	91	619
Local extinctions caused by invasive alien microbes	2	12	0	15
All taxa	**282**	**755**	**176**	**1,215**

extinctions have occurred in all realms, but most extinctions have been documented in the terrestrial (62.1 per cent) realm, followed by the marine (23.2 per cent) and inland waters (14.5 per cent) realms **(Table 4.3)**. Overall, invasive alien animals have caused the most local extinctions of native species (vertebrates 51.0 per cent, invertebrates 32.5 per cent) in the terrestrial realm; whereas invasive alien plant species (15.3 per cent) and microbes (1.2 per cent) have caused fewer local extinctions **(Table 4.3)**. In contrast, in the marine realm, invasive alien invertebrates (47.5 per cent) and plants (34.8 per cent) are more often documented to be the cause for local extinctions than invasive alien vertebrates (17.0 per cent; **Table 4.3**).

Invasive alien vertebrates dominate the list of species causing local extinctions, e.g., *Felis catus* (cat), *Rattus Rattus* (black rat), *Rattus exulans* (Pacific rat), *Vulpes vulpes* (red fox), and *Capra hircus* (goats), but also the marine fish *Pterois volitans* (red lionfish). Ants also often lead to local extinctions, particularly species such as *Linepithema humile* (Argentine ant), *Anoplolepis gracilipes* (yellow crazy ant), and *Solenopsis invicta* (red imported fire ant). Plants that frequently lead to local extinctions are *Caulerpa cylindracea* (green algae) and *Pontederia crassipes* (water hyacinth). Microbes are less frequently implicated in local extinctions; pathogens that have caused local extinctions are *Batrachochytrium dendrobatidis* (chytrid fungus), *Austropuccinia psidii* (myrtle rust), *Ceratocystis platani* (canker stain of plane), *Cryphonectria parasitica* (blight of chestnut), the *Haplosporidium nelsoni* (MSX oyster pathogen), and *Morator aetatulas* (sacbrood virus) that affects honeybee larvae.

Global extinctions

Where invasive alien species caused global extinctions **(Box 4.4)**, the impacted native species often had a restricted spatial distribution with immutable borders. Thus, species endemic to islands, mountain ranges, or isolated lakes and river systems seem to be particularly at risk of global extinction caused by invasive alien species. Examples include *Boiga irregularis* (brown tree snake), which caused the local extinction and serious reduction of populations of most of the Guam's resident 25 bird species (Wiles *et al.*, 2003), leading to the global extinction of *Myiagra freycineti* (Guam flycatcher). Several global extinctions were attributed to the invasive alien *Euglandina rosea* (rosy predator snail), a predatory snail native to Central America and southern United States of America, introduced to many pacific islands to control *Lissachatina fulica* (giant African land snail) (Gerlach *et al.*, 2021). This terrestrial predatory snail led to the global extinction of several, mostly tree-inhabiting island-endemic snails of the genus *Partula* (Coote & Loève, 2003). The invasive alien *Rattus Rattus* (black rat) has been documented as the only cause of the global extinctions of *Nesoryzomys darwini* and *Nesoryzomys indefessus* (rice rats) endemic to the protected areas of the Galapagos Islands (Tirira & Weksler, 2017, 2019).

Box 4 4 Global extinctions caused by invasive alien species.

Invasive alien species have a range of impacts on nature which can ultimately lead to the global extinction of native species. The IUCN Red List synthesized information about species extinctions as well as their associated threats, which provides a basis to study the impact of invasive alien species in terms of extinctions. The IUCN Red List documented 327 animal and plant species as globally extinct or extinct in the wild with invasive alien species mentioned as one of the causes of extinctions, with an additional 205 species that are considered possibly extinct (average 50 years since the last specimen was seen). Invasive alien species are the only cause attributed to 16 per cent of all species extinctions documented in that database (K. G. Smith, 2020). Invasive alien species are also categorized as a significant contributing factor (i.e., having caused significant decline to the majority of the species' ranges) in nearly 60 per cent of extinctions, while in the remaining cases the role of invasive alien species as driver of extinctions is unknown and most likely minor compared to other drivers of change. By focusing on species extinctions in which the primary cause has been identified, invasive alien species are by far the most frequently mentioned driver. Note that most of the species that have gone extinct due to invasive alien species were also harmed by wildlife exploitation and/or cultivation and those threats are likely to act in combination on insular species (Leclerc *et al.*, 2018).

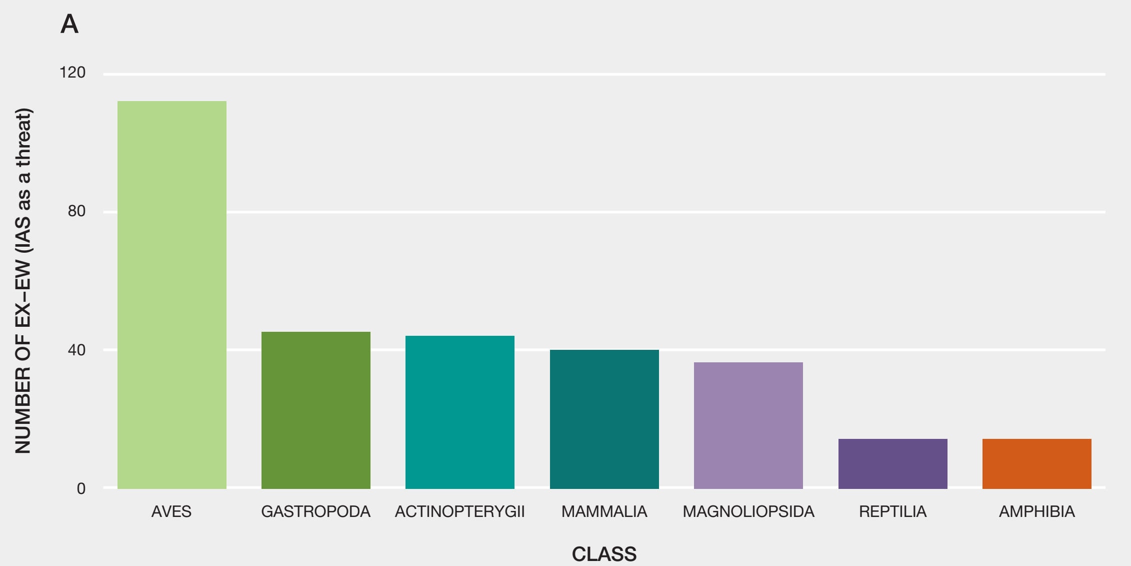
Among the extinctions in which invasive alien species are categorized as a significant cause (n=186), the overwhelming majority occurred on oceanic or continental islands (90 per cent). The risk of extinctions was greater on islands presumably because the species had reduced geographical range (**Glossary**), small population size, and reduced pressure from native predators compared to continental species (J. G. Cox & Lima, 2006; **Boxes 4.6**, **4.7**). For instance, naïve island birds, that have never encountered mammalian predators such as rodents and *Felis catus* (cat), are particularly vulnerable (Dueñas *et al.*, 2021; Medina *et al.*, 2011; Whitworth *et al.*, 2013; **Figure 4.4**). Extinction hotspots where invasive alien species are documented as the main cause are located in the Asia-Pacific region (73 per cent), followed by the Americas (15 per cent) and Africa (14 per cent). Vertebrates (62.4 per cent), invertebrates (26.3 per cent) and plants (11.3 per cent) suffered most extinctions as a consequence of invasive alien species, with birds (74 species) most vulnerable. (H. P. Jones *et al.*, 2008; Spatz *et al.*, 2014; Szabo *et al.*, 2012). This threat continues to the present (Butchart, 2008; Dueñas *et al.*, 2021).

Box 4 4

Figure 4 4 **Examples of extinct birds due to invasive alien cats on islands.**

On islands, *Felis catus* (cat) has caused the extinction of *Anthornis melanocephala* (Chatham bellbird; left), *Cyanoramphus novaezelandiae erythrotis* (Macquarie Island parakeet; middle), and *Microgoura meeki* (Choiseul pigeon; right). Photo credits: Lynx Edicions, Jan Wilczur (left); Norman Arlott (middle); John Cox (right) – Copyright.

The number of mollusc extinctions documented by the IUCN Red List may underestimate the role of invasive alien species. A recent re-evaluation attributed at least 134 inland waters species extinctions exclusively to the introduction of the notorious predatory alien snail, *Euglandina rosea* (rosy predator snail; Régnier *et al.*, 2009). This species was intentionally introduced in the 1950s to the 1970s as a biological control agent for *Lissachatina fulica* (giant African land snail) in many Pacific islands. It also feeds on other snails and consequently a third of the species within the snail family Partulidae (Gastropoda) in the Pacific Islands are now extinct, the rest being at risk of extinction (Gerlach, 2016). Inland waters fish are also particularly affected by invasive alien species, with 8 native fish species documented as extinct worldwide due predominantly to interactions with various invasive alien fish and an additional 37 fish extinctions attributed to invasive alien species as one of several causes. Note that the same taxa are affected (i.e., birds, gastropods, fishes, mammals and angiosperms) when considering all the extinct species where invasive alien species are cited as one of the causes, but not necessarily the main one **(Figure 4.5)**.



Box 4 4

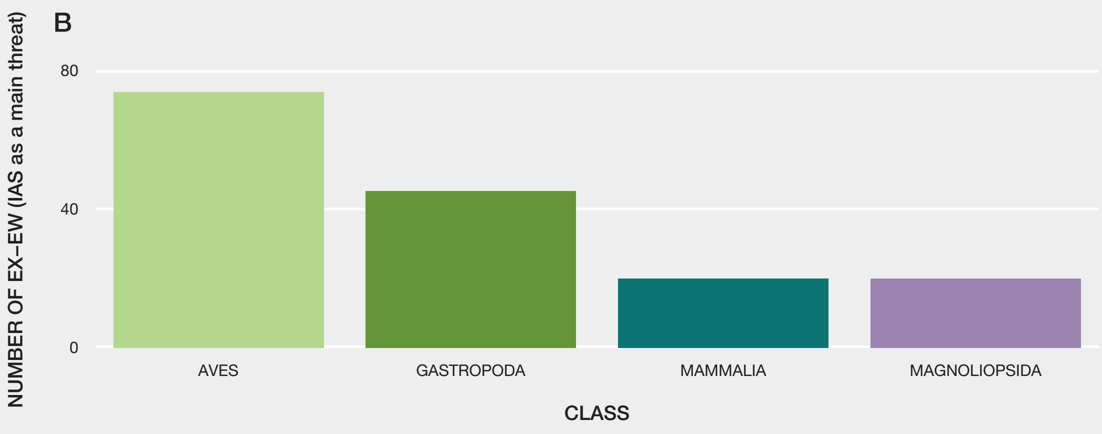


Figure 4 5 **Number of extinct species, including extinct in the wild (EX-EW), by classes.**

These bar plots only show classes represented by at least 10 extinct species, with A) all extinct species (EX-EW) where invasive alien species (IAS) are cited as one of several causes (but not necessarily the main one) and B) all extinct species (EX-EW) where invasive alien species (IAS) are considered as the main cause of extinction. A data management report is available at https://doi.org/10.5281/zenodo.5762737

At least 44 invasive alien species are implicated in the 186 extinctions documented by the IUCN as caused by invasive alien species, with rodents and *Felis catus* (cat) involved in more than a third of all extinctions. The large majority of the invasive alien species are represented by alien mammals that were responsible for the extinctions of native birds and mammals, while most of the amphibians are threatened by *Batrachochytrium dendrobatidis* (chytrid fungus; Pounds *et al.*, 2006).

Despite the fact that the IUCN Red List is recognized as the world's most comprehensive information source on the global conservation status of species, it should be emphasized that the attribution of factors driving extinctions relies on evidence from published literature (including peer review) in addition to expert opinion and thus is subject to data gaps in observations and to some level of uncertainty (IUCN, 2022; Salafsky *et al.*, 2008). The scientific literature frequently lacks specific

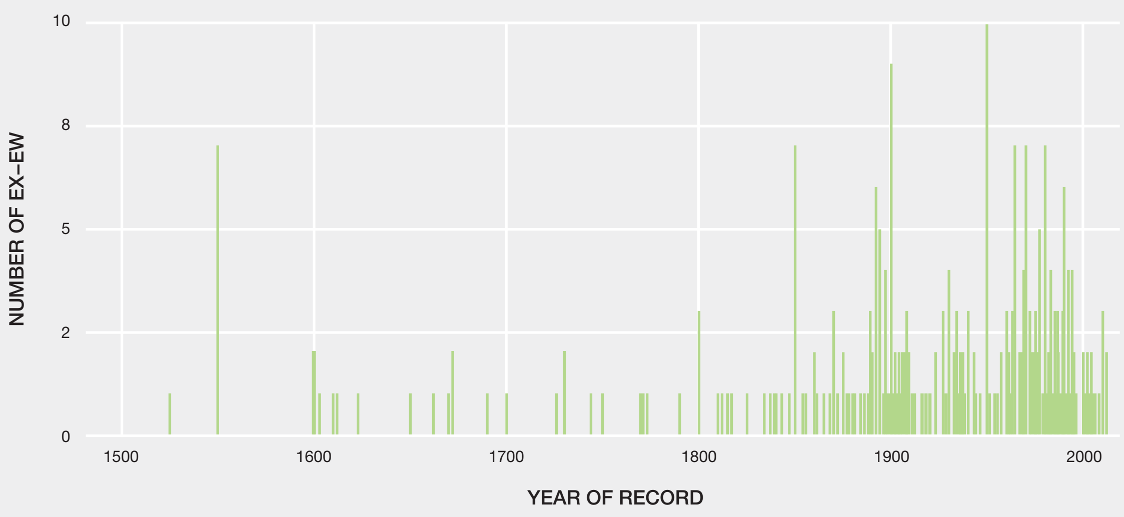


Figure 4 6 **Number of records of species extinct or extinct in the wild (EX-EW) since 1500, with their last seen date indicated.**

Information was available for 276 extinct species. Data source: IUCN (2021).

Box 4 4

information on already extinct species. This is particularly true for extinctions that occurred before the 1950s for which clear information is scarce (**Figure 4.6**; see also Sayol *et al.*, 2020). A recent systematic review, of manipulative experimental or comparative observational (before-after; control-invaded plots; BACI design; Kumschick *et al.*, 2015), on current extinction threat confirmed findings from the IUCN Red List on the strong impact of invasive alien species on threatened species (Dueñas *et al.*, 2021). Yet, the most prevalent threats across near-threatened and threatened species worldwide are overexploitation and agricultural activity (Maxwell *et al.*, 2016). It is thus crucial to emphasize that the number of extinctions or the number of species at risk of extinction are not the only reliable metrics to study the impacts of invasive alien species. The IPBES Global Assessment of Biodiversity and Ecosystem Services synthesized 11 biodiversity indicators including local species richness, mean body length and the IUCN Red List indices (Purvis *et al.*, 2019). Using those indices, the relative importance of invasive alien species may vary across ecosystems, taxa, and measure of biodiversity (Bellard *et al.*, 2022). As a consequence, the contribution of invasive alien species to explain the current biodiversity crisis should be carefully discussed considering the specific context, taxa, and metrics to avoid oversimplifications.

Global extinctions due to invasive alien species are not restricted to islands. *Batrachochytrium dendrobatidis* (chytrid fungus) is a pathogen of a wide range of amphibians and can be found in the Americas, Africa, Western Europe, South-East Asia and Australia (M. C. Fisher & Garner, 2020). Its origin is still disputed, but it has been widely transported and introduced by humans, initially probably with the global use of *Xenopus laevis* (African clawed frog) for medicinal purposes starting in the 1950s (Kay & Peng, 1992). *Batrachochytrium dendrobatidis* has contributed to the severe global decline of amphibians generally (M. C. Fisher & Garner, 2020), and has caused the global extinction of several native harlequin toads of the genus *Atelopus* from the mountains of Central America (La Marca *et al.*, 2005), most likely because climate warming increased the habitat suitability for the pathogen (Pounds *et al.*, 2006; **Box 4.5**).

Global extinctions due to invasive alien species have also occurred in aquatic realms. As an example, *Lates niloticus* (Nile perch) was introduced from its native range in Lake Albert to Lake Victoria to improve local fisheries but led to the global extinction of many cichlid fish species endemic to Lake Victoria (Goudswaard *et al.*, 2008). It remains controversial whether the introduction of *Lates niloticus* has profited local fishermen **(Box 4.10)**.

No global extinctions due to invasive alien species were documented in the marine realm; this might be partly because immutable dispersal borders are less frequent in the marine realm. Yet, one should also take into consideration that it is far more difficult to document impacts and their causality in marine environments due to accessibility challenges (Ojaveer *et al.*, 2015).

Box 4 5 **Invasive alien species impacts can worsen when interacting with other drivers of change.**

Invasive alien species occur in interaction with other major drivers of biodiversity change, such as climate change, land- and sea-use change, pollution and over exploitation of natural resources (IPBES, 2019a; **Chapter 3**, **section 3.5**). Interactions may be classified as additive, antagonistic or synergistic with examples of all outcomes evident from studies on the interactions between invasive alien species and other drivers. Research on multiple drivers of biodiversity change is challenging, with drivers operating at different temporal and spatial scales (Bonebrake *et al.*, 2019). Additionally, the interdisciplinary skills and resources required to study multiple drivers may not be available in all regions of the world. A recent metanalysis, assessing 458 cases from 95 published studies (with 74 of these being laboratory or mesocosm experiments) on individual and combined effects of drivers of change on invasive alien species, demonstrated that synergistic interactions were documented for more than 25 per cent of the studies (Lopez *et al.*, 2022). However, it is notable that in most cases the impacts of invasive alien species were not exacerbated by the other drivers, but the combined impacts of the other drivers with invasive alien species were typically no worse than the impacts from invasive alien species alone. Documented synergistic interactions mostly lead to the deterioration of ecosystems (Lopez *et al.*, 2022). There are several studies that have provided evidence on the synergistic effects of invasive alien species and other drivers of biodiversity change, here we highlight four examples of these phenomena **(Figure 4.7)**.

Climate change and invasive alien plants increase fire frequency and intensity

In a scenario of climate change, where vast areas of the Earth will not only be warmer but drier, and the number of lightning events is expected to increase, invasive alien plants may worsen the situation by adding additional highly flammable fuel (Aslan & Dickson, 2020; Turbelin & Catford, 2021). For example, *Pinus* spp. (pine) invading grasslands and forests

Box 4 5

of South America may increase fire intensity and frequency (Cóbar-Carranza *et al.*, 2014; Paritsis *et al.*, 2018; **Chapter 1**, **Box 1.4**). Similar effects have been documented in the South African fynbos (O'Connor & van Wilgen, 2020).

Climate change and alien mosquitoes threaten the health of humans and animals

For some vector-borne diseases, climate change may increase the range and the density of the invasive alien species vector (**Glossary**) with profound implications for human health. For example, *Anopheles* spp. (mosquitoes) that carry malaria have been documented to advance into higher latitudes of the Americas and Europe (Tjaden *et al.*, 2018; Brugueras *et al.*, 2020). Climate change has also been implicated in the rapid decline of amphibians as it interacts with the invasive pathogenic *Batrachochytrium dendrobatidis* (chytrid fungus), which may explain population reductions and even extinctions (J. M. Cohen *et al.*, 2019).

Pollution and invasive organisms can transform water bodies

Change in nutrient levels due to pollution can increase populations of aquatic invasive alien species in inland waters and marine ecosystems reducing native species diversity (Crooks *et al.*, 2011). Some aquatic invasive alien plants, such as *Pontederia crassipes* (water hyacinth), thrive in highly polluted eutrophicated habitats, worsening the consequences for local fisheries, infrastructure and transport (Villamagna & Murphy, 2010; Kleinschroth *et al.*, 2021).

Hunting and invasive alien vertebrates can bring populations of native species in islands or forests to extinction (steep population reductions)

A combined effect of hunting and invasive alien predators may cause faster local reductions in populations of endangered vertebrates such as birds, amphibians and mammals (e.g., New Zealand birds in Innes *et al.*, 2010). Furthermore, hunting can also be a source of new introductions of game animals (Carpio *et al.*, 2017), but in some cases, hunting may also be an effective tool to control invasive alien species (Jean Desbiez *et al.*, 2011).

Interactions amongst three or more drivers including invasive alien species

Conceptualization and quantification of impacts of the interaction of three or more drivers is a highly complicated endeavour (e.g., birds in Doherty *et al.*, 2015; bats in Frick *et al.*, 2020; deer and earthworms in Frelich *et al.*, 2006). However, evidence suggests that measures including research could address these complex interactions to concurrently reduce the threats of multiple drivers on biodiversity. Evidence suggests that while invasive alien species can exacerbate the impacts of other drivers of biodiversity change, the impacts of invasive alien species are generally no worse when acting in combination with other drivers of change such as climate change, pollution, over exploitation or land and sea use change (Lopez *et al.*, 2022). Indeed, managing biological invasions locally contributes to reducing the threat of multiple drivers of change (**Chapter 3**, **section 3.5**; **Chapter 5**, **section 5.6.1.3**).

Figure 4 7 **Species affected by multiple drivers of change, including invasive alien species.**

Atelopus toads threatened by extinction due to *Batrachochytrium dendrobatidis* (chytrid fungus) which grows better due to climate change; water hyacinth covering large parts of eutrophic tropical lakes due to increased nitrogen pollution. Photo credits: Brian Gratwicke, WM Commons – CC BY 2.0 (left) / NickLubushko, WM Commons – CC BY 4.0 (right).

Case study: Avian botulism, a probable synergistic impact of alien species and climate warming in the North American Great Lakes

In the North American Great Lakes, several alien species are considered to contribute to recurring mass die-offs of waterfowl by transmitting botulin toxin. Filtration activities of Ponto-Caspian dreissenid mussels (*Dreissena polymorpha* (zebra mussel) and *Dreissena rostriformis bugensis* (quagga mussel) introduced to the Great Lakes in the 1980s) increase light transparency in the water and consequently promote excessive summer growth of macrophytes and benthic macroalgae (Vanderploeg *et al.*, 2002). Later in the summer,

Box 4 5

the decomposing biomass of this vegetation, combined with elevated water temperatures resulting from climate change generated hypoxic conditions, favouring outbreaks of a rare cryptogenic (**Glossary**) strain of botulism bacteria, *Clostridium botulinum* Type-E (Chun *et al.*, 2013). The bacteria are then filtered by dreissenid mussels, which concentrate the toxic cells in their tissues. The mussels and other contaminated benthic invertebrates subsequently transfer the toxin to *Neogobius melanostomus* (round goby), a benthic predatory Ponto-Caspian fish introduced to the Great Lakes region, that is itself a common prey item for piscivorous native waterfowl such as loons and gulls (Essian *et al.*, 2016; **Figure 4.8**). Thus, the combination of alien species and increased temperatures through climate change promotes the proliferation and transfer of botulinum toxin to higher trophic levels, creating a new contaminant pathway (**Glossary**) that has caused the mortality of tens of thousands of waterfowls in the Great Lakes nearly every year over the past two decades (Essian *et al.*, 2016; Yule *et al.*, 2006).

Figure 4 8 **Dead waterbirds on a beach on Georgian Bay, Lake Huron, October 2011.**

Waterbirds mass die-off is attributed to botulism, which is considered to proliferate due to the presence of invasive alien species. Photo credit: Rogers Media Inc. – CC BY 4.0.

Reduction of population sizes

Many invasive alien species that are not documented to cause local extinctions can still reduce the population size of native species. The database of impacts developed through this chapter shows that 21.8 per cent (766) of the studied invasive alien species have caused a total of 4,282 local population declines in native species, which represents 36.6 per cent of the documented negative impacts on nature. While many invasive alien plants have not been documented causing local extinctions, they frequently cause declines in native species populations. Such invasive alien plants include the terrestrial forbs *Reynoutria japonica* (Japanese knotweed), *Impatiens glandulifera* (Himalayan balsam), *Carpobrotus* spp. (iceplant), *Eragrostis lehmanniana* (Lehmann lovegrass), or *Acacia longifolia* (golden wattle) and *Robinia pseudoacacia* (black locust). Other invasive alien species that have been documented as frequently causing at least local population declines are *Acridotheres tristis* (common myna) or *Bubalus bubalis* (Asian water buffalo), and *Phytophthora ramorum* (sudden oak death).

Invasive alien plants have the highest number of species causing impacts on nature, followed by invertebrates and vertebrates. Comparatively few invasive alien microbes are documented causing impacts on nature **(Figure 4.9A)**. By contrast, more invasive alien animals (vertebrates and invertebrates) cause high magnitude impacts, i.e., local extinctions, both in terms of the number of species causing impacts **(Figure 4.9A)** and the number of reports **(Figure 4.9B)**. Local extinctions are less frequently caused by plants, and microbes are rarely documented to have caused local extinctions. Invasive alien plants also have the highest number of species causing impacts on ecosystem properties, population declines and reductions in individual performance of native species.

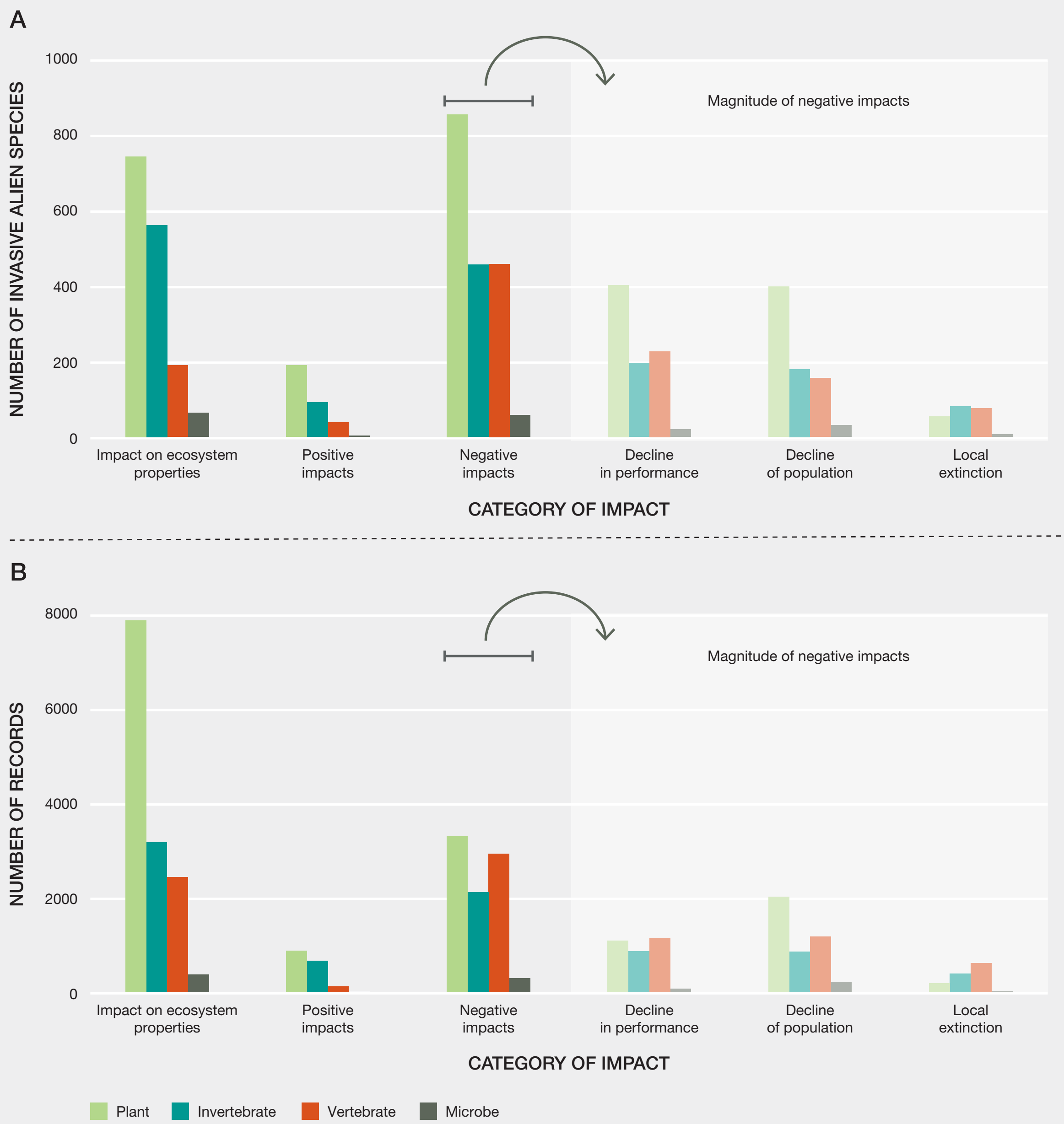


Figure 4 9 **Number of A) invasive alien species causing impacts on nature and B) number of impact records by direction, magnitude and taxonomic group.**

Negative and positive impacts relate to the consequences for native species, while impacts on ecosystem properties are not assigned a direction. Negative impacts are subdivided for each invasive alien species into the maximum documented negative impact on a native species globally (lighter box on the right). Note that the same invasive alien species could have a maximum negative impact on a native species, impacts on ecosystem properties and positive impacts on native species. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Impacts on native species by invasive alien vertebrates are more frequent than impacts on ecosystem properties **(Figure 4.9A)**, while, in contrast, invasive alien invertebrates more frequently cause negative impacts on ecosystem properties (e.g., abiotic soil or water characteristics; **Box 4.3**) than on native species. Invasive alien plants and microbes have similar frequencies of negative impacts on ecosystem properties and native species.

Invasive alien plants have the highest numbers of species (both in absolute numbers and percentages; **Figure 4.9A**) and reports **(Figure 4.9B)** of positive impacts on nature globally. There are more invasive alien invertebrates than vertebrates causing positive impacts, but microbe species are very rarely documented in this respect.

4.3.1.1 Islands *versus* mainland

Islands, particularly smaller sized and more isolated islands, with higher rates of endemism, suffered greater impacts on nature than mainland regions. Following the introduction of various suites of alien predators and competitors through millennia of human settlement, the severity and rate of extinction has varied due to geomorphology, composition of the native biota and that of the introduced invasive alien species, and lifestyle and technology of human settlers (e.g., hunting-gatherering, husbandry, agriculture) (Wood *et al.*, 2017). Smaller islands which tend to support smaller native populations in easily accessible habitats, are more susceptible to impacts from invasive alien species through predation or habitat loss, leading to greater rates of extinction and other negative impacts (Duncan & Blackburn, 2007).

Impacts on islands represent 4,820 (19.9 per cent) of the total number of impacts on nature documented in published studies (**Boxes 4.6** and **4.7**). Negative impacts on native species on islands are far more frequent (40.5 per cent) than positive impacts (4.5 per cent; **Figure 4.10**).

There is no clear difference in the proportion of negative and positive impacts and impacts on ecosystem properties between island and non-island locations **(Figure 4.10)**. However, negative impacts of high magnitude, i.e., local extinctions, are more frequently documented from islands than from non-island locations (9.2 per cent vs. 4.0 per cent; **Figure 4.10**).

On islands, globally, 87 invasive alien species have caused 445 local extinctions. They are most often caused by mammals, such as *Rattus* spp. (rats), *Capra hircus* (goats), *Mus musculus* (house mouse), *Felis catus* (cat), but also other vertebrates such as *Anas platyrhynchos* (mallard) and *Boiga irregularis* (brown tree snake). Ants have also often led to local extinctions on islands, particularly from species such as *Anoplolepis gracilipes* (yellow crazy ant), *Wasmannia auropunctata* (little fire ant), *Linepithema humile* (Argentine ant), and *Pheidole megacephala* (big-headed ant). Invasive alien plants are much less often causing local extinctions on islands, but there are some reports, as for example from *Vachellia nilotica* (gum arabic tree).

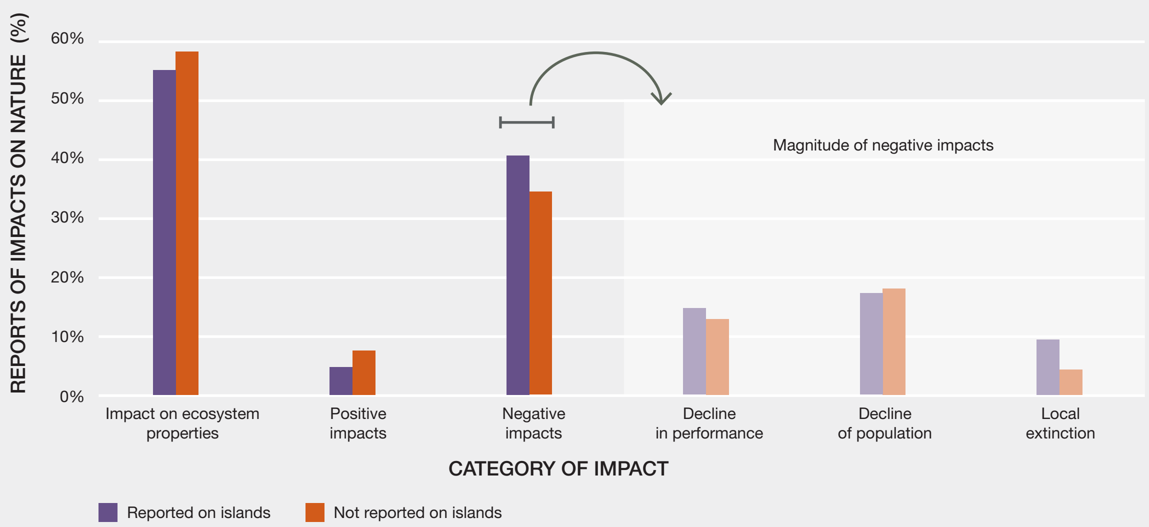


Figure 4 10 **Percentage of reports of impacts on nature on islands vs. locations that were either not on an island (mainland) or unknown.**

For each of the three impact categories, ecosystem properties, positive and negative impacts, the percentages sum to 1. Negative impacts are split into their impact magnitudes in the shaded box on the right-hand side. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Box 4 6 Hawaiian extinctions – the birds and the bees and much else.

"The islands now contain more endangered species per square mile than anywhere else in the world" (Cabin, 2013).

Hawaii's isolation in remote Oceania resulted in a unique terrestrial biota, distinguished by the diversity of endemic forms derived from a small number of ancestral immigrations (Wagner *et al.*, 1990; Imada, 2012). Zooarchaeological and archaeobotanical studies reveal that the arrival of humans about 1500 years ago induced catastrophic ecosystem changes (Allen, 1984, 1989; Steadman, 1995). The Polynesian settlers transported their domesticates, including dogs, pigs, chickens, and the synanthropic *Rattus exulans* (Pacific rat; Crabtree, 2016). Their direct and indirect impacts, through habitat loss, fragmentation, and predation by the introduced mammals, caused losses of many species, and are continuing into the present (Vitousek & Walker, 1989; G. W. Cox, 1999; Staples *et al.*, 2000; Staples & Cowie, 2001; Vorsino *et al.*, 2014; MacLennan, 2017). Pollen analysis reveals that following Polynesian settlement Hawaii's lowland forests were reduced to a landscape dominated by opportunistic shrubs and grasses. Athens (1997, 2009) considers the Pacific rat as the underlying cause of lowland forest collapse. Only remnant populations sequestered in the least accessible habitats remain of the unique Hawaiian biota.

Olson and James (1982) estimate that the extinction of more than half the endemic avifauna of the Hawaiian Islands, including two thirds of endemic flightless, ground-nesting land birds and burrowing sea-birds, occurred between the initial human settlement and arrival of Europeans. At least 71 species or subspecies died out before the nineteenth century, and 24 have gone extinct since. The remaining populations are declining or are in danger of extinction (T. K. Pratt *et al.*, 2006). Cats, rodents, and mongoose have been the major extinction cause for ground nesting birds (G. W. Cox, 1999). Introduced avian diseases, such as *Avipoxvirus* spp. (avian pox virus) and *Plasmodium relictum* (avian malaria), have also led to the decline of the endemic Hawaiian avifauna (Warner, 1968; van Riper III *et al.*, 1986, 2002; Samuel *et al.*, 2015). Avian malaria is uniquely transmitted by *Culex quinquefasciatus* (southern house mosquito), introduced before 1830 (Fonseca *et al.*, 2000; LaPointe, 2000), the larvae of which are found in high densities in low- and mid-elevation forests, degraded by the foraging behaviour of *Sus scrofa* (feral pig; Lapointe, 2008).

Interactions amongst native pollinators and plants are considered to be important for long-term sustainability of natural island ecosystems (S. K. Walsh *et al.*, 2019). Loss of plant-pollinating birds has disrupted plant-pollinator relations and led to plant extinctions, e.g., 31 species of bird-pollinated bellflowers, Campanulaceae, have gone extinct. Introduced *Zosterops japonicus* (Japanese white-eye) outcompetes native birds for insects and nectar (G. W. Cox, 1999), providing a replacement pollinator for *Freycinetia arborea* (ie'ie vine), once pollinated by extinct or endangered native birds (P. A. Cox, 1983; P. A. Cox & Elmqvist, 2000). *Pheidole megacephala* (big-headed ant) and *Linepithema humile* (Argentine ant) threaten the once abundant native *Hylaeus* spp. (yellow-faced bees; Perkins, 1913; Lach, 2005; Magnacca & King, 2013; Sahli *et al.*, 2016; Plentovich *et al.*, 2021), of which half are now extinct, threatened, or extremely rare (Magnacca, 2007). These native bees exhibit high pollinator fidelity to native species, whereas *Apis mellifera* (European honeybee) also pollinates invasive alien plant species (E. E. Wilson *et al.*, 2010; Miller *et al.*, 2015; Aslan *et al.*, 2016). Endemic Hawaiian honeycreepers and insects are the most important pollinators of the iconic native tree *Metrosideros polymorpha* ('ōhi'a lehua) culturally important to the native Hawaiians (Kānaka Maoli), and a keystone species of the Hawaiian rainforest (Cortina *et al.*, 2019). 'Ohi'a lehua trees have been in rapid decline since the 1960s (Akashi & Mueller-Dombois, 1995; Gruner, 2004) from infection by introduced *Ceratocystis* fungi (Keith *et al.*, 2015; Barnes *et al.*, 2018; Heller *et al.*, 2019), likely transmitted by introduced *Xyleborus* spp. (ambrosia beetles; K. Roy *et al.*, 2020).

As many as 90 per cent of the 750 recognized species of land snails are extinct; the once speciose Amastridae, comprising more than 300 species endemic to Hawaii, are currently reduced to 10 species (Lydeard *et al.*, 2004). Many land snails were annihilated by *Euglandina rosea* (rosy predator snail) introduced in a failed attempt to control the previously introduced *Lissachatina fulica* (giant African land snail; G. W. Cox, 1999). The native freshwater gastropods, too are threatened with extinction or are much reduced in range (Christensen *et al.*, 2021).

Box 4 7 Impacts of invasive alien species on nature in Antarctica, Antarctic and sub-Antarctic Islands.

Few alien species have established on the Antarctic continent and its surrounding islands south of 60°S, within the Antarctic Treaty region, on land or in the many waterbodies on the continent, which vary greatly in salinity (Frenot *et al.*, 2005; Cavicchioli, 2015; Hughes *et al.*, 2015; McGeoch *et al.*, 2015; Bergstrom, 2021). Though alien crabs, mussels and tunicates have been documented from Antarctic coasts, none have established populations (López-Farrán *et al.*, 2021). Currently, alien taxa are limited to the Antarctic Peninsula and adjacent islands, mostly to areas under strong human pressure, such as the vicinity of research stations and sites attractive to tourists (Znój *et al.*, 2017). They include the recently documented

Box 4 7

mussel *Mytilus* cf. *platensis* (Cárdenas *et al.*, 2020), a chironomid midge *Eretmoptera murphyi*, for which direct evidence on impact on native species is lacking (J. C. Bartlett *et al.*, 2020), and the fly *Trichocera maculipennis* (winter crane fly), which is yet to be explicitly confirmed as established in the natural environment (Remedios-De León *et al.*, 2021).

Impact studies were conducted solely on the invasive alien grass *Poa annua* (annual meadowgrass) in Antarctica (Hughes *et al.*, 2015; Baird *et al.*, 2019; **Figure 4.11**). Experimental and modelling studies suggest that the invasive grass *Poa annua* could have impacts on the only two vascular plant species indigenous to the Antarctic: the grass *Deschampsia antarctica* (Antarctic hair grass) and the forb *Colobanthus quitensis* (Antarctic pearlwort) (Molina-Montenegro *et al.*, 2019). Observational and experimental data of co-occurrence of vascular plant species in the Antarctic Peninsula revealed that *Deschampsia antarctica* facilitates the presence of *Poa annua* and may impact its introduction and spread (Atala *et al.*, 2019).

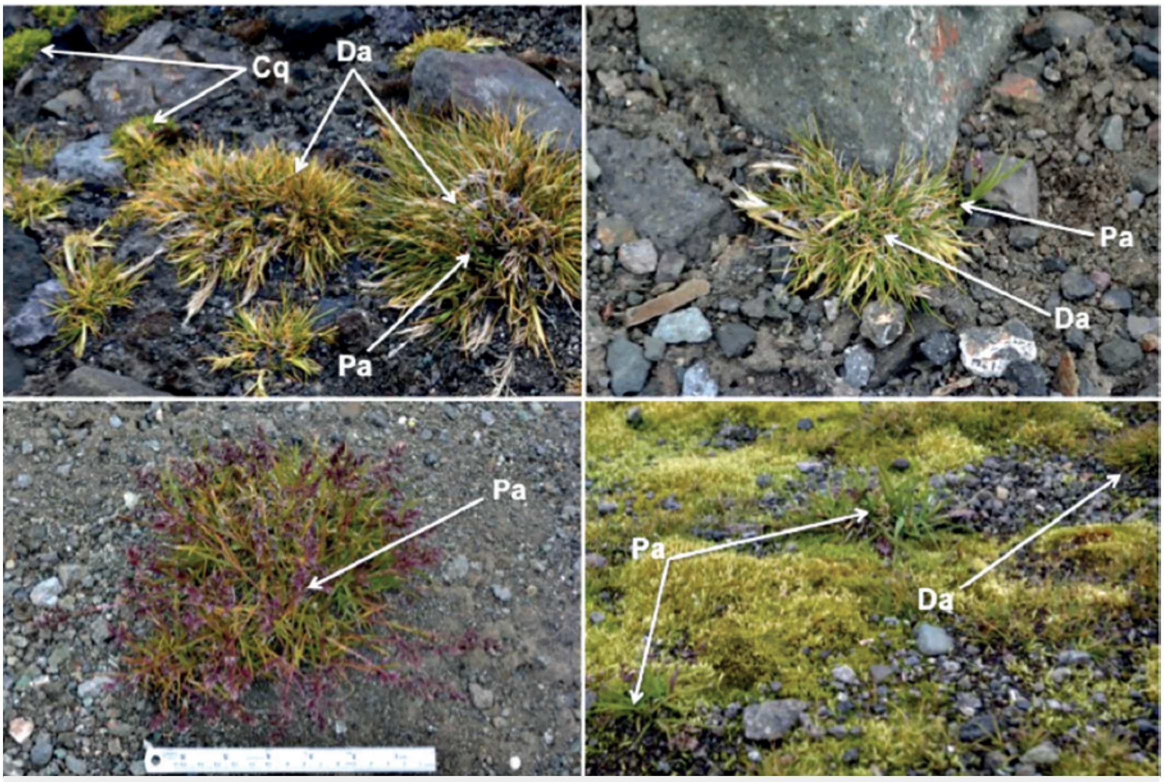


Figure 4 11 ***Poa annua* (annual meadowgrass), the only invasive alien species with documented impacts on the Antarctic continent.**

Source: Molina-Montenegro *et al.* (2019), https://doi.org/10.3897/neobiota.51.37250, under license CC BY 4.0.
Da: *Deschampsia antarctica* (Antarctic hair grass)
Pa: *Poa annua* (annual meadowgrass)
Cq: *Colobanthus quitensis* (Antarctic pearlwort)

Due to their size and extreme isolation, the Southern Cold Temperate Islands (e.g., Tristan da Cunha, New Zealand Shelf Islands), and the sub-Antarctic Islands (e.g., South Georgia and the South Sandwich Islands, Crozet Islands, Heard Island), are taxonomically and functionally depauperate, and thus, vulnerable to synanthropic introductions (Dawson *et al.*, 2022; Frenot *et al.*, 2005; Hughes *et al.*, 2020).

Terrestrial mammals, absent prior to their introductions, have posed, and still pose, the most severe threats to the islands' biodiversity, ecosystem structure and landscape (McGeoch *et al.*, 2015). Introduced feral herbivores, such as *Bos taurus* (cattle), *Rangifer tarandus* (reindeer), *Ovis aries* (sheep), and *Oryctolagus cuniculus* (rabbits), have all had significant impacts on the vegetation of the islands to which they have been introduced (Vogel *et al.*, 1984; Chapuis *et al.*, 1994, 2004; Whinam *et al.*, 2014). In some cases, direct impacts led to indirect ones: on Macquarie Island, eradication of cats was followed by increasing rabbit population, resulting in island-wide ecosystem effects, altered vegetation structure, impacting burrow-nesting seabirds; on Ile Verte rabbit eradication enabled the rapid expansion of the invasive alien *Taraxacum officinale*

Box 4 7

(dandelion), and impacted both native, burrowing seabird prey populations and their predator, *Stercorarius skua* (great skua) (Chapuis *et al.*, 2004; Scott & Kirkpatrick, 2008; Bergstrom *et al.*, 2009; Whinam *et al.*, 2014; Brodier *et al.*, 2011; Houghton *et al.*, 2019). On South Georgia and the South Sandwich Islands, reindeer caused the replacement of indigenous grasses with the introduced grazing-tolerant *Poa annua*, a poor food for the indigenous *Hydromedion sparsutum* (tussac beetle), and thus indirectly contributed to its decline (Chown & Block, 1997). *Felis catus* (cat) has had major impacts on burrowing and other seabird populations on the islands to which they were introduced (Frenot *et al.*, 2005). Significant recovery of some populations followed their removal (Dilley *et al.*, 2017; Brooke *et al.*, 2018), and was tempered, in some cases, by the increase in populations and impacts impacts of *Felis catus*' invasive alien prey (e.g., *Oryctolagus cuniculus* (rabbits); Bergstrom *et al.*, 2009). Invasive alien rodents such as *Rattus Rattus* (black rat), *Rattus norvegicus* (brown rat), and *Mus musculus* (house mouse), have had significant impacts on invertebrate populations, to the point of extirpation in some cases (V. Le Roux *et al.*, 2002; McClelland *et al.*, 2018; J. C. Russell *et al.*, 2020). Rats have caused the near disappearance of terrestrial birds, such as *Anthus antarcticus* (South Georgia pipit; now recovering following rat eradication), and are also thought to be responsible for declines in the abundance of burrowing seabird species (Pye & Bonner, 1980; Jouventin *et al.*, 2003; H. P. Jones *et al.*, 2008; Dilley *et al.*, 2018). Mice were documented preying on naïve chicks and adults of several albatross and burrowing petrel species (M. G. W. Jones & Ryan, 2010; Dilley *et al.*, 2016, 2018; C. W. Jones *et al.*, 2019).

Invasive alien predatory beetles have led to substantial declines in the abundance of their preferred invertebrate prey on Kerguelen Island (Lebouvier *et al.*, 2011; Houghton *et al.*, 2019). Although many plant species have become invasive on the sub-Antarctic islands, impacts were only quantified for a few of them: *Agrostis stolonifera* (creeping bentgrass) reduces the abundance of indigenous plants and alters arthropod community structure on Marion Island (Gremmen *et al.*, 1998), and the widespread *Poa annua* outcompetes indigenous plants for space and resources, especially in coastal areas disturbed by seals and penguins (Haussmann *et al.*, 2013; L. K. Williams *et al.*, 2018).

4.3.1.2 Protected areas

Invasive alien species frequently impact protected areas around the world (Carlton *et al.*, 2019; Foxcroft, Richardson, *et al.*, 2013; Galil, 2017; Macdonald *et al.*, 1988). A report of the Working Group on Nature Reserves, associated with the SCOPE programme on the "Ecology of Biological Invasions" concluded that all nature reserves, except those in Antarctica, appear to have invasive species (Usher, 1988). Invasive alien species have reduced, or have the potential to reduce, the viability of protected areas to provide refugia for native species, habitats and the ecosystem services that they sustain (Foxcroft, Richardson, *et al.*, 2013). Impacts on nature in protected areas represent 19.3 per cent (4,673) of the total number of impacts on nature documented in published studies. Negative impacts on native species in protected areas are far more frequent (33.2 per cent) than positive impacts (6.3 per cent; **Figure 4.12**).

There is no clear difference in the proportion of negative and positive impacts and impacts on ecosystem properties inside and outside protected areas **(Figure 4.12)**, although declines of native populations seem to be higher inside protected areas than outside (20.3 per cent vs. 17.1 per cent; **Figure 4.12**). Thus, protected areas are not sheltered by their protection status from impacts of invasive alien species.

In protected areas, globally, 53 invasive alien species have caused 240 local extinctions of native species. *Rattus Rattus* (black rat) is by far the most frequent invasive alien species causing local extinctions in protected areas, but other mammals such as *Capra hircus* (goats), *Felis catus* (cat), or *Sus scrofa* (feral pig) have also been documented multiple times. Local extinctions of a native species have not only been restricted to the terrestrial realm, but have also occurred in the marine realm caused multiple times by invasive alien species such as *Sporobolus* spp. (cordgrass), *Caulerpa taxifolia* (killer algae), *Halophila stipulacea* (halophila seagrass), *Kappaphycus alvarezii* (elkhorn sea moss), or *Mytilus galloprovincialis* (Mediterranean mussel), and in inland waters caused by *Pontederia crassipes* (water hyacinth). The microbial pathogenic *Batrachochytrium dendrobatidis* (chytrid fungus) has also caused local amphibian extinctions in protected areas.

4.3.1.3 Mechanisms

The mechanisms through which invasive alien species can impact native species are classified according to the EICAT (IUCN, 2020; **Box 4.2**). They include competition; predation; hybridization; transmission of disease; parasitism; poisoning or toxicity; bio-fouling or other direct physical disturbance; grazing, herbivory or browsing; chemical, physical, structural impact on ecosystems; and indirect impacts through interactions with other species (**Box 4.3** for definitions). Note that standards for classification of impact mechanisms have only been defined for negative impacts at the time of developing this assessment report; thus, positive impacts are not discussed here with respect to their different mechanisms (but see Vimercati *et al.*, 2022).

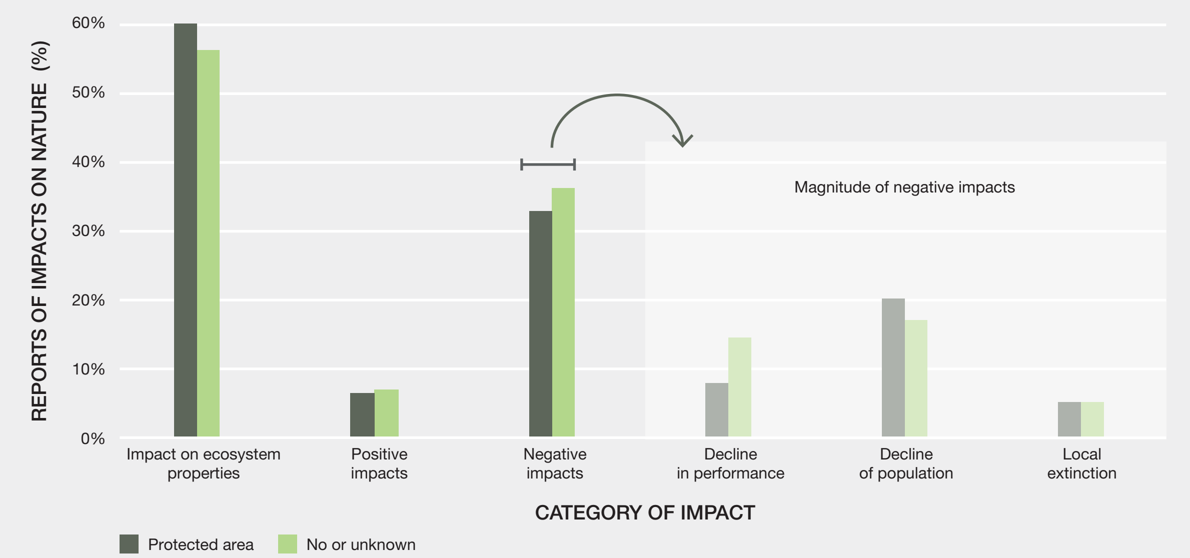


Figure 4 12 **Percentage of reports on impacts on nature in protected areas vs. locations that were either not in a protected area or unknown.**

For each of the three impact categories, ecosystem properties, positive and negative impacts, the percentages sum to 1. Negative impacts are split into their impact magnitudes in the shaded box on the right-hand side. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Table 4 4 **Distribution of mechanisms across invasive alien species taxa and realms.**

Examples are not meant to be representative but highlight invasive alien species which have caused local extinctions of native populations.

Plants: Invertebrate: Vertebrate: Microorganisms: Inland waters: Marine: Terrestrial:

Mechanism	Main taxa	Realms	Examples of invasive alien species
Competition	Plants, Invertebrate	Inland waters, Marine, Terrestrial	*Linepithema humile* (Argentine ant), *Solenopsis invicta* (red imported fire ant), *Caulerpa cylindracea* (green algae)
Predation	Invertebrate, Vertebrate	Inland waters, Marine, Terrestrial	*Felis catus* (cat), *Vulpes vulpes* (red fox), *Pterois volitans* (red lionfish), *Lates niloticus* (Nile perch)
Hybridization	Plants, Vertebrate	Inland waters, Marine, Terrestrial	*Anas platyrhynchos* (mallard), *Ambystoma tigrinum* (tiger salamander), *Sporobolus densiflorus* (denseflower cordgrass)
Transmission of disease	Invertebrate, Vertebrate	Inland waters, Terrestrial	*Faxonius limosus* (spiny-cheek crayfish), *Canis lupus familiaris* (dogs)
Parasitism	Invertebrate, Microorganisms	Inland waters, Marine, Terrestrial	*Philornis downsi* (avian vampire fly), *Batrachochytrium dendrobatidis* (chytrid fungus), *Haplosporidium nelsoni* (MSX oyster pathogen)
Toxicity	Plants, Vertebrate	Marine, Terrestrial	*Caulerpa taxifolia* (killer algae), *Rhinella marina* (cane toad)
Biofouling	Plants, Invertebrate	Inland waters, Marine	*Kappaphycus alvarezii* (elkhorn sea moss), *Carijoa riisei* (branched pipe coral), *Dreissena polymorpha* (zebra mussel)
Herbivory	Invertebrate, Vertebrate	Inland waters, Marine, Terrestrial	*Capra hircus* (goats), *Carcinus maenas* (European shore crab), *Ctenopharyngodon idella* (grass carp)
Ecosystem	Plants, Invertebrate	Inland waters, Marine	*Pontederia crassipes* (water hyacinth), *Caulerpa cylindracea* (green algae), *Mytilus galloprovincialis* (Mediterranean mussel)
Indirect	Invertebrate, Vertebrate	Inland waters, Marine, Terrestrial	*Dreissena* spp. (zebra/quagga mussel), *Pterois volitans* (red lionfish), *Bromus tectorum* (downy brome)

Invasive alien species affect ecosystem properties and native species through all mechanisms leading to varying degrees of magnitude of impact. The occurrence of each mechanism is not evenly distributed across taxa and realms **(Table 4.4)**. While examples of most mechanisms can be found for all invasive alien species and across realms, some mechanisms are more commonly associated with some taxa and within specific realms. For example, most reports of hybridization of an alien species with a native species relate to invasive alien vertebrates and plants, while invasive alien invertebrates and microbes seem to hybridize much less frequently with native species **(Table 4.4)**. Likewise, transmission of diseases seems to be less frequent in the marine realm than in terrestrial and inland waters systems, and toxicity less frequent in the inland waters **(Table 4.4)**.

Impacts on nature by invasive alien species are most often caused by changes to ecosystem properties (26.8 per cent) and competition (23.7 per cent), followed by predation (18.4 per cent) and herbivory (12.3 per cent), i.e., direct trophic interactions. Indirect mechanisms (disease transmission and interactions with other species) only account for 8.7 per cent of all records, but this might be partly due to indirect interactions being less often studied or overlooked due to their complexity. Across all types of mechanism, changes to ecosystem properties alongside impacts of lower magnitude are more often documented than high magnitude impacts (**Box 4.2** for impact magnitudes).

Local extinctions are most often caused through hybridization (19.5 per cent), followed by impacts through predation (11.8 per cent) and direct physical interactions/ biofouling (7.2 per cent) **(Figure 4.13)**. Invasive alien species that are most commonly responsible for extinctions through hybridization are *Anas platyrhynchos* (mallard) and *Ambystoma tigrinum* (tiger salamander), followed by *Cervus nippon* (sika) and *Oreochromis niloticus* (Nile tilapia). Invasive alien species most frequently causing extinctions through predation are the terrestrial vertebrates *Vulpes vulpes* (red fox), *Felis catus* (cat), *Rattus* spp. (rats) and *Boiga irregularis* (brown tree snake). In the marine realm, *Pterois volitans* (red lionfish) and *Paralithodes camtschaticus* (red king crab) have caused frequent local extinctions through predation on native species. Terrestrial invertebrates are documented to have less frequently caused local extinctions through predation, and the most frequently documented species involved in local extinctions are *Anoplolepis gracilipes* (yellow crazy ant), *Pheidole megacephala* (big-headed ant), and *Euglandina rosea* (rosy predator snail).

4.3.1.4 Affected native species

Most documented impacts of invasive alien species are on native plants (8,472 reports), closely followed by native invertebrates (6,253 reports) and vertebrates (5,144 reports), but the number of invasive alien species affecting native plants is much higher than for the other taxa **(Figure 4.14A)**.

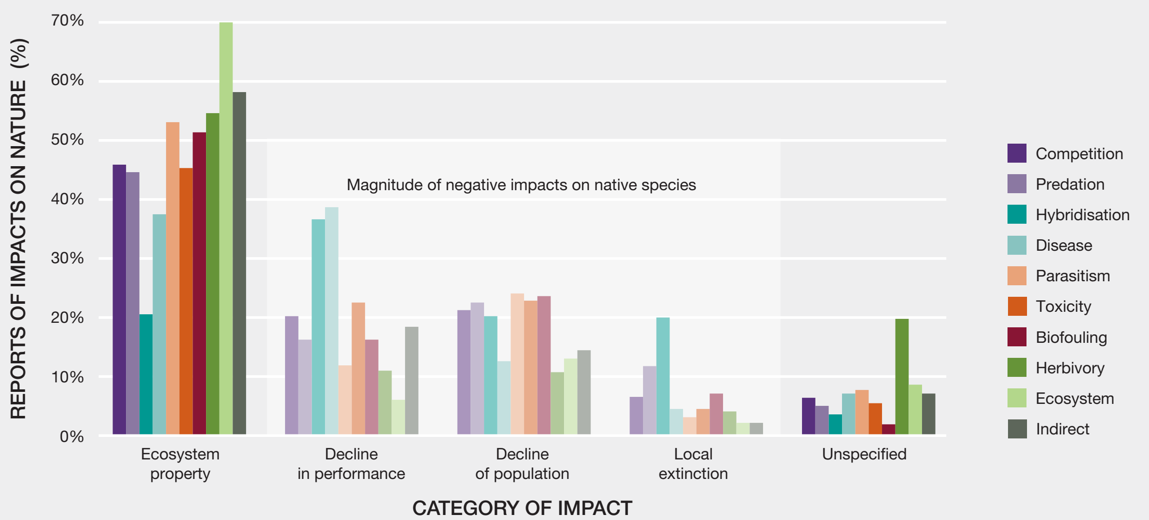


Figure 4 13 **Percentage of reports of impacts on nature through different mechanisms.**

Percentage of reports (y axis) for different categories of impact on nature: ecosystem property, native species and unspecified (x axis). For each mechanism, the percentages sum to 1. Negative impacts on native species at three different magnitudes are highlighted in the shaded inset box. No mechanisms are defined for positive impacts, and these are not considered here. Mechanisms are: competition, predation, hybridization, transmission of disease, parasitism, poisoning/toxicity, bio-fouling or other direct physical disturbance, grazing/herbivory/browsing, chemical, physical, structural impact on ecosystem, and indirect impacts through interactions with other species (IUCN, 2020). A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Native microbes are rarely documented to be impacted by invasive alien species, which is most likely to be a research gap; only 1.2 per cent of documented impacts of invasive alien species relate to native microbes. There were 1,215 reports of local extinctions of native species due to 218 invasive alien species **(Table 4.5A; Figure 4.14A)**.

The number of invasive alien species positively affecting native species is less than 10 per cent for native plants, but increases to about 15 per cent for both vertebrates and invertebrates, and is highest for native microbes (over 25 per cent; **Figure 4.14A**), which can have higher abundance in soil communities dominated by invasive alien

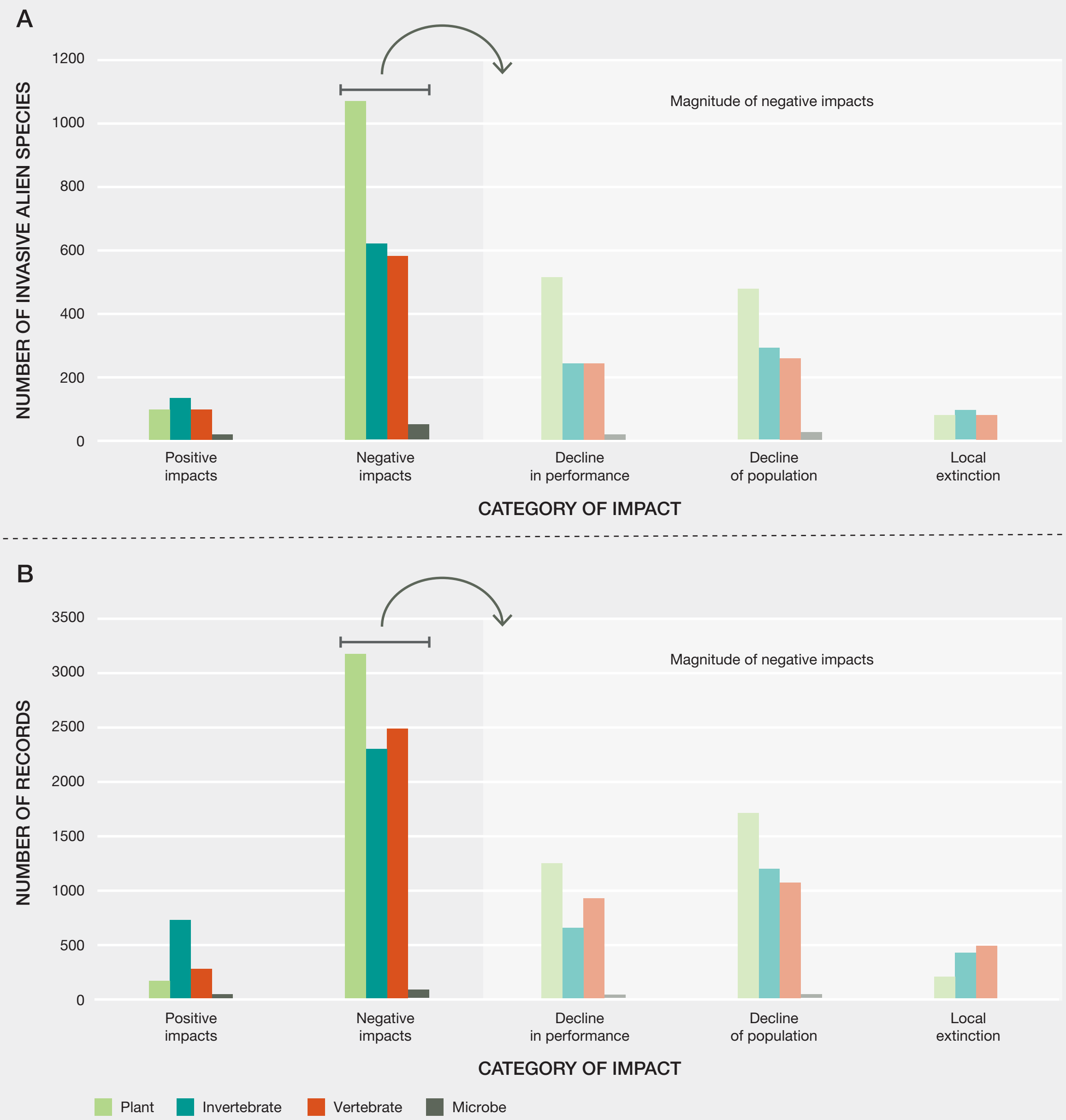


Figure 4 14 **Number of invasive alien species A) and number of impact records B) affecting native taxa by direction and magnitude.**

Number of records (y axis) for different categories of impacts (x axis). Negative and positive impacts relate to the consequences for native species, while ecosystem impacts are not assigned a direction. Negative impacts in A) are subdivided for each invasive alien species into the maximum documented negative impact on a native species globally (shaded boxes on the right-hand side). Note that the same invasive alien species could have a maximum negative impact on native species from different taxa, impacts on ecosystem properties and positive impacts on native species. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069 plants (de Souza *et al.*, 2018). The proportion of positive to negative impacts remains similar regarding the number of impacts documented except for invertebrates where the proportion of positive impacts rises to almost 25 per cent **(Figure 4.14B)**. Negative impacts of invasive alien species occur most often within the same taxonomic group **(Table 4.5A)**, i.e., invasive alien plants most often negatively impact native plants, and invasive alien vertebrates most often impact native vertebrates. However, this pattern does not hold for invasive alien microbes, which predominantly negatively impact plants (plant pathogens). The overall pattern is slightly different in positive impacts **(Table 4.5B)**, where invasive alien plants predominantly positively affect native invertebrates, either by providing a food source or habitat structure, while invasive alien invertebrates and vertebrates mostly positively affect native species in their own taxonomic group.

Documented local extinctions caused by invasive alien species mostly affect populations of native vertebrates, followed by native invertebrates and plants **(Figure 4.14B)**. Invasive alien species can also cause evolutionary responses in native species **(Box 4.8)**.

Table 4 5 **Records of negative and positive impacts of invasive alien species on native taxa.**

Number of negative A) and positive B) impacts on native taxa caused by invasive alien species, documented by the chapter impact database. Impacts within the same taxonomic groups in alien and native taxa are italicized. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

A) Negative impacts of invasive alien species on native taxa

Native taxa	Invasive alien species: Plants	Invertebrates	Vertebrates	Microorganisms
Plants	*2,542*	795	462	283
Invertebrates	701	*1,437*	399	38
Vertebrates	336	418	*2,027*	52
Microorganisms	84	12	5	*1*

B) Positive impacts of invasive alien species on native taxa

Native taxa	Invasive alien species: Plants	Invertebrates	Vertebrates	Microorganisms
Plants	*276*	29	20	1
Invertebrates	404	*452*	15	1
Vertebrates	99	129	*59*	--
Microorganisms	49	8	2	*--*

Box 4 8 Invasive alien species as drivers of rapid evolutionary change in native species.

Invasive alien species often dramatically alter habitat conditions for native species. As dominant community members, they may also act as novel resources (e.g., prey) for, or threats (e.g., predators) to, native species. These changes may lead to rapid evolutionary responses in native species (Carroll, 2007; G. W. Cox, 2004; J. J. Le Roux, 2021). Generally, changes caused by invasive alien species to host plants or food resources, the bio-physical environment, mortality and reproductive rates in native species, and competitive interactions, facilitate rapid adaptive evolution in native species (J. J. Le Roux, 2021). Non-adaptive shifts in the trait values of native species may also occur in response to invasive alien species, e.g., when they hybridize with invasive alien species (Todesco *et al.*, 2016). Alien species also often undergo rapid evolution throughout the biological invasion process which, in turn, may exacerbate their impacts (J. J. Le Roux, 2021).

The strength of selection pressure on native species brought about by invasive alien species, or vice versa, partly depends on how often these species interact and the levels of eco-evolutionary experience they share with one another (Saul *et al.*, 2013; Saul & Jeschke, 2015). Eco-evolutionary experience describes the historical exposure of species to biotic interactions, highlighting the role of preadapted traits in driving the biological invasion success of alien species (Saul *et al.*, 2013; Heger *et al.*, 2019). Therefore, the eco-evolutionary experience of alien species will determine how quickly their populations become widespread, as well as the form and strength of their interactions with native species (Carroll, 2007). Selection pressures on both alien and native species are expected to be strong when native species share moderate to high levels of eco-evolutionary experience with invasive alien species, e.g., when native invertebrates colonize invasive alien plants that are closely related to their native host plants (Carroll *et al.*, 2005). Native species lacking eco-evolutionary experience with experienced invasive alien species are also likely to experience strong selection, whereas alien species that share little eco-evolutionary experience with conditions in the new environment may fail to establish (J. J. Le Roux, 2021).

Direct impacts

Direct interactions between native and invasive alien species may cause rapid evolution in the native species for them to avoid, exploit, or co-exist with invasive alien species (J. J. Le Roux, 2021). Soapberry bugs in the subfamily Serinethinae provide classic examples of such impacts. As their name suggests, these bugs are herbivores of plants in the family Sapindaceae (Carroll & Loye, 2012). Given this eco-evolutionary experience, soapberry bugs have colonized various invasive Sapindaceae species in many parts of the world, often resulting in rapid evolutionary responses in these insects. For example, invasive balloon vines (genus *Cardiospermum*) have been repeatedly colonized by native *Leptocoris* soapberry bugs in Australia (Andres *et al.*, 2013; Carroll *et al.*, 2005) and South Africa (Foster *et al.*, 2019).

Balloon vines carry their seeds in inflated capsules, an adaptation to insect predators with piercing mouth parts. In Australia, the native soapberry bug *Leptocoris tagalicus* rapidly evolved longer proboscides (or "beaks") to increase its feeding efficiency on invasive *Cardiospermum grandiflorum* (Carroll *et al.*, 2005). Similarly, in South Africa host shifts by native *Leptocoris mutilatus* onto two invasive balloon vines (*Cardiospermum halicacabum* and *Cardiospermum grandiflorum*), not only led to the evolution of longer beaks, but also to the formation of genetically-distinct host races (Foster *et al.*, 2019). In the South-western United States, *Jadera haematoloma* (red-shouldered bug) shifted from its native *Cardiospermum* balloon vine host onto the invasive *Koelreuteria elegans* (goldenrain tree; Carroll & Boyd, 1992). In this instance, the bug was confronted with flatter seedpods on its new host plant, leading to the rapid evolution of shorter beaks (Carroll & Boyd, 1992; **Figure 4.15**).

Invasive alien species may also have significant evolutionary consequences when they act as novel resources for native species. On the one hand, native predators may experience strong selection to increase their ability to capture or consume palatable invasive prey or, conversely, to avoid toxic ones. Native Australian predators of invasive *Rhinella marina* (cane toad) illustrate how quickly such evolutionary impacts can happen. *Rhinella marina* produce a potent cocktail of defensive toxins that differs in its chemical constituents from the toxins produced by native Australian anurans (Daly *et al.*, 1987). Therefore, most Australian predators lack eco-evolutionary experience with cane toad toxins. Despite this, invasive cane toads are frequently attacked and consumed by native predators, presumably because of their superficial resemblance to Australian frogs. The amount of toxin produced by cane toads varies throughout their life cycle, with older and larger toads being more poisonous than younger and smaller ones (Hayes *et al.*, 2009). Snakes are gape-limited, and the size of their heads thus determines the size of prey they can consume. The evolution of smaller head (or gape) size is therefore likely to occur in toad-naïve predators, because those that can swallow larger toads would be removed from the breeding population. Morphological data from four Australian snake species, spanning a period of 80 years since the arrival of cane toads, partly support this hypothesis. As predicted, Phillips and Shine (2004) found two species, *Pseudechis porphyriacus* (red-bellied black snake) and *Dendrelaphis punctulatus* (common tree snake), to have evolved smaller heads since the arrival of cane toads in Australia. By contrast, *Hemiaspis signata* (swamp snake) and *Tropidonophis mairii* (common keelback snake) did not display any evolutionary responses to invasive *Rhinella marina*. *Hemiaspis signata* already have unusually small heads, making them incapable of ingesting toads large enough to kill them (Phillips *et al.*, 2003). While *Tropidonophis mairii* have normal-sized heads, their Asian ancestry, and thus historical exposure to poisonous toads, likely provided them with sufficient eco-evolutionary experience to tolerate their poisoning (Phillips & Shine, 2004).

Box 4 8

Figure 4 15 **Native species may experience strong selection when they utilize abundant invasive alien species as novel food sources.**

Invasive alien balloon vines in the genus *Cardiospermum* have been repeatedly colonized by native soapberry bugs. Shown here is the perennial balloon vine (*Cardiospermum grandiflorum*, main picture) in South Africa that has been colonized by the native bug *Leptocoris mutilatus* (inserted picture). In order to feed on balloon vine seeds more efficiently, some soapberry bug populations have rapidly evolved longer mouthparts. Photo credit: Johannes Le Roux – CC BY 4.0.

Invasive alien predators may also cause strong evolutionary responses in native species. In Lombardy, Italy, invasive alien *Procambarus clarkii* (red swamp crayfish) is established in waterbodies throughout the region (Ficetola *et al.*, 2011). Prior to its arrival, tadpoles of different populations of the native *Rana latastei* (Italian agile frog) exhibited pronounced variation in development time, depending on water temperature; this variation disappeared following the arrival of the red swamp crayfish (Melotto *et al.*, 2020). Within 14 years of the crayfish's introduction, tadpoles of the frog developed significantly faster in invaded ponds than in uninvaded ponds, irrespective of whether these were in foothill or lowland areas. These evolutionary responses likely occurred to reduce the frog's exposure to crayfish predation by allowing earlier metamorphosis and, remarkably, occurred over just 3-6 frog generations. Invasive alien species may also act as mutualists for native species (J. J. Le Roux *et al.*, 2020).

Indirect impacts

Invasive alien species may also create indirect evolutionary pressures on native species by changing abiotic and/or biotic conditions in ways that indirectly affect the fitness of native species (J. J. Le Roux, 2021). For example, along coasts of South-eastern Australia, the invasive seaweed *Caulerpa taxifolia* (killer algae), reduces water flow rates and causes anoxic sediment conditions, leading to increases in the abundance of large phytoplankton species (Gribben *et al.*, 2009; McKinnon *et al.*, 2009). These changes are thought to underlie the rapid evolution of longer and broader shells in the native mussel, *Anadara trapezia* (Sydney cockle), presumably for this mollusc to cope with altered sediment conditions and food resources (J. T. Wright *et al.*, 2012). While indirect evolutionary impacts are likely common, they are hard to predict and quantify (Berthon, 2015).

Hybridization between closely related native and invasive alien species is frequently documented. Genetic introgression, i.e., when hybrid offspring backcross to one or both parental species, can dilute native gene pools and purge them of locally adapted genotypes. This may ultimately lead to the extinction of native populations (Rhymer & Simberloff, 1996; Todesco *et al.*, 2016). A well-studied example is *Anas platyrhynchos* (mallard). This highly successful invasive alien species hybridizes with several native duck species around the world (Stephens *et al.*, 2020). Many of these hybrids are fertile and subsequent introgression has been documented in many instances (e.g., Rhymer *et al.*, 1994; Mank *et al.*, 2004; Stephens *et al.*, 2020). For example, in New Zealand, introgressive hybridization has led to the virtual elimination of *Anas superciliosa superciliosa* (New Zealand grey duck; Lavretsky, 2020).

The ecological consequences of the evolutionary impacts of invasive alien species

How biological invasions change and shape the evolutionary trajectories of native species is highly context-dependent and hard to predict, making inferences of long-term ecological

Box 4 8

and biodiversity impacts difficult. In the worst-case scenario, evolutionary impacts may lead to the extinction of native species. *Occidryas editha* (Edith's checkerspot butterfly) is a particularly dramatic example; it occurs in western United States and utilizes a narrow range of short-lived annual host plants. At Schneider's Meadow, Nevada, *Occidryas editha* rapidly demonstrated a rapid evolution of preference for invasive alien *Plantago lanceolata* (ribwort plantain; Singer & Parmesan, 2018). Selection for this host shift was strong, because, unlike the butterfly's native host plants, ribwort plantain provided its larvae with food year-round (Singer & Parmesan, 2018). Subsequent changes in land-use led to the recovery of grassland vegetation and the smothering of *Plantago lanceolata* plants, creating microclimatic conditions that were unsuitable for larval development. *Occidryas editha* was unable to switch back to their original native host plants at Schneider's Meadow and the local population died out (Singer & Parmesan, 2018). *Occidryas editha* is highly sedentary and therefore this local extinction likely led to the permanent loss of unique phylogenetic history.

Rapid evolution in native species may also reduce the impacts they experience from invasive alien species. For instance, in the United States, invasive *Alliaria petiolata* (garlic mustard) impacts native plants by interfering with their mycorrhizal fungal mutualists *via* strong allelopathy (Lankau, 2012). Invasive populations of *Alliaria petiolata* also rapidly evolved higher levels of allelopathy (Lankau *et al.*, 2009). Lankau (2012) found native *Pilea pumila* (clearweed) to have evolved tolerance to *Alliaria petiolata* and the ability to maintain high levels of mycorrhization in invaded areas. This suggests that co-evolutionary dynamics exist between the invasive alien species and native species.

Invasive alien species may facilitate speciation. A classic example is *Sporobolus anglicus* (common cordgrass), the descendant lineage of hybrids between invasive *Sporobolus alterniflorus* (smooth cordgrass) and native *Sporobolus maritimus* (small cordgrass) (Gray *et al.*, 1991). In another example, invasive *Lonicera* honeysuckles in North America acted as new host plants shared between two native tephritid fruit flies, *Rhagoletis mendax* (blueberry fruit fly) and *Rhagoletis zephyria* (snowberry fruit fly) (Schwarz *et al.*, 2005). These host shifts led to the breakdown of historical ecological barriers (i.e., utilization of distinct native host plants) between the two fly species and the establishment of a genetically-distinct hybrid fly lineage that is reproductively isolated from both parent species (Schwarz *et al.*, 2005).

The evolutionary responses of native species to invasive alien species likely ramify throughout entire communities and ecosystems, yet our understanding of such broad-scale impacts remains limited. For example, native insects may experience altered parasitoid loads when a native parasitoid evolves preference for a new and abundant invasive alien insect host. This may lead to community-wide changes in insect-host interactions.

4.3.2 Documented impacts of invasive alien species on nature by realm

4.3.2.1 Patterns of negative and positive impacts of invasive alien species on nature in the terrestrial realm

The database of impacts developed through this chapter includes more than 10,000 impacts on nature in the terrestrial realm, implicating 1,588 invasive alien species. Among these documented impacts, 76 per cent (6,638) can be considered as negative impacts, 17 per cent (1,498) as neutral and only 7 per cent (651) as positive impacts. Negative impacts in terrestrial habitats are caused by a total of 1,186 invasive alien vertebrates, invertebrates, plants or microbes.

The chapter impact database highlights that the vast majority of negative impacts caused by invasive alien species on nature are in terrestrial habitats (70 per cent). This bias towards terrestrial impacts is most likely a consequence of the rate at which humans have transported and introduced alien species through time (Seebens *et al.*, 2017), but may also reflect a bias in terrestrial research over inland waters and marine research.

Mechanisms and magnitude of negative impacts

Among all the species negatively impacting terrestrial habitats, more than half (52 per cent, 619 species) cause decline in the performance of native species, almost half (45 per cent, 530 species) cause decline of local native populations, and some (9 per cent, 105 species) cause local extinctions of native species. The magnitude of negative impacts is context-dependent; some invasive alien species have impacts of different magnitudes in different invaded habitats. The highest numbers of invasive alien species causing decline in the performance of native species are found in boreal forests and woodlands, cultivated areas and tropical and subtropical dry and humid forests but the highest numbers of invasive alien species causing impact of greater magnitude, that is, population decline of native species and local extinction, are found in tropical and subtropical dry and humid forests and temperate and boreal forests and woodlands **(Table 4.6)**. By contrast, tundra and high mountain habitats and deserts and xeric shrublands are the ecosystems with lowest records of negative impacts caused by invasive alien species on nature.

Table 4 6 **Number of negative impacts of invasive alien species on nature in the terrestrial realm, by unit of analysis.**

Number of invasive alien species (IAS) and records of negative impacts on nature in the terrestrial realm for each unit of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Unit of analysis	Maximum impact on native species					
	Decline in performance		Decline of population		Local extinction	
	IAS	Records	IAS	Records	IAS	Records
Tropical and subtropical dry and humid forests	144	404	160	586	46	282
Temperate and boreal forests and woodlands	129	389	159	728	33	196
Mediterranean forests, woodlands and scrub	57	158	73	251	17	115
Tundra and high mountain habitats	8	22	31	60	4	5
Tropical and subtropical savannas and grasslands	63	222	97	326	30	104
Temperate grasslands	188	381	106	366	14	66
Deserts and xeric shrublands	20	50	31	64	9	85
Urban/Semi-urban	167	311	54	159	9	15
Cultivated areas (incl. cropping, intensive livestock farming etc.)	239	479	84	173	16	24

Impacted units of analysis

Temperate and boreal forests and woodlands, and tropical and subtropical dry and humid forests are the most impacted units of analysis in the terrestrial realm, with, respectively, 1,313 and 1,272 negative impacts **(Table 4.6)**. Particularly, these are the habitats with the highest reports both of decline of native local populations and local extinctions caused by invasive alien species. For example, in the United States, *Lumbricus terrestris* (lob worm) can be found in temperate and boreal habitats and has caused the reduction of plant-species richness and changed plant communities in mature forests (Holdsworth *et al.*, 2007). Some vertebrates are also invading these forests, for example *Castor canadensis* (North American beaver; IPBES, 2018a) has invaded temperate forests, but also grasslands and peatlands in southern Argentina and Chile, causing several negative ecological and economic impacts (Duboscq-Carra *et al.*, 2021; Gaiarin & Durham, 2016; Valenzuela *et al.*, 2013). *Castor canadensis* is considered an ecosystem engineer due to the magnitude of the changes it produces in the riparian environments – it invades by building dams which affect nutrient cycling and soil properties, chemistry, biodiversity, morphology, flow and water dynamics of rivers and streams. *Castor canadensis* builds its dams by cutting down trees, degrading riparian forests. Associated with these modifications that it generates in the environment, *Castor canadensis* facilitates the invasions of other alien species, both aquatic and terrestrial (Gaiarin & Durham, 2016; Valenzuela *et al.*, 2013). The invasion of *Castor canadensis* has also economic impacts: the costs associated with the invasion of *Castor canadensis* in Argentina are estimated to be around 66.56 million United States dollars (US$; Duboscq-Carra *et al.*, 2021). Tropical and subtropical humid and dry forests are amongst the most extensive ecosystems in South America and are being impacted by several invasive alien species that mostly originated from tropical areas in Asia and Africa (IPBES, 2018a). Some examples of invasive alien plant species in these habitats are *Pinus patula* (Mexican weeping pine) in Colombia (GISP, 2005); *Artocarpus heterophyllus* (jackfruit) in Brazil (Fabricante *et al.*, 2012); *Ligustrum lucidum* (broad-leaf privet) in Argentina (Hoyos *et al.*, 2010), and *Acacia mangium* (brown salwood) in French Guiana (Delnatte & Meyer, 2012) and Brazil (Heringer *et al.*, 2019). According to published studies, tundra and high mountain habitats and deserts and xeric shrublands not only have the lowest number of invasive alien species causing negative impact but also have the lowest records of negative impacts among all terrestrial habitats **(Table 4.6)**. For example, *Ulex europaeus* (gorse), one of the most impactful invasive alien plant species in the world (Global Invasive Species Database, 2010), is invading several high Andean regions, altering the structure of plant communities which negatively affects the composition of birds (Amaya-Villarreal & Miguel Renjifo, 2010).

Invasive alien taxa most often documented causing negative impacts on nature in the terrestrial realm

Invasive alien plant species are responsible for almost half (45 per cent) of all the negative documented impacts on nature in the terrestrial realm (e.g., **Box 4.9**), followed by invasive alien vertebrates (27 per cent), invertebrates (23 per cent) and microbes (5 per cent). Several invasive alien plants cause negative impacts at different levels of ecological organization, from individual species populations to native plant and animal communities to ecosystems as a whole. For example, the shrub *Lantana camara* (lantana) has adverse impacts on native understorey shrubs and herbaceous plants diversity, and affects the vegetation community composition by reducing seedling recruitment of vertebrate-dispersed seeds (Dobhal *et al.*, 2010; Kohli *et al.*, 2006; Prasad, 2010; Raghubanshi & Tripathi, 2009; Sundaram *et al.*, 2012). It also increases soil nitrogen that may further favour its proliferation (Sharma, 2011). This shrub is unpalatable, and replaces native palatable herbs and reduces available forage for wild ungulates (Prasad, 2010; G. Wilson *et al.*, 2014). Physical changes of large extensions of invaded habitats by *Lantana camara* can change habitat use of large mammals such as elephants (G. Wilson *et al.*, 2013). In forests, increased density of this shrub is correlated with a decrease in bird diversity, with certain guilds (canopy and insectivorous birds) being more adversely affected than others (Aravind *et al.*, 2010). *Lantana camara* also alters fire regimes by increasing the fuel load of invaded forests, leading to more intense and severe fires (Hiremath & Sundaram, 2005; Kohli *et al.*, 2006; Sundaram *et al.*, 2012; Tireman, 1916). Furthermore, in temperate and boreal regions of north-western Europe, *Picea sitchensis* (Sitka spruce) is assessed to be among the highest-risk alien species in Norway (Norwegian Biodiversity Information Centre, 2018; Sandvik *et al.*, 2020) as well as in Great Britain and Ireland (Dehnen-Schmutz *et al.*, 2022). *Picea sitchensis* severely changes ecological conditions across a significant proportion of the habitat area of red-listed habitats such as coastal *Calluna*-heathlands and coastal mires, with knock-on impacts on red-listed plants, birds and other species linked to these habitats (Hinderaker & Nielsen, 2022; Norwegian Biodiversity Information Centre, 2018; Øyen & Nygaard, 2020; Saure *et al.*, 2013, 2014). These heathlands are now critically endangered throughout their range in western Europe, due to compound threats involving land-use change, nutrient pollution, and invasive alien species (IPBES, 2018b). Other examples of invasive alien plant species with multiple simultaneous impacts are species belonging to the Pinaceae family. Pinaceae comprises some of the most invasive tree species and at least 20 species of the genus *Pinus* are considered to be invasive in at least one region of the southern hemisphere (Richardson & Rejmanek, 2004). These invasive alien species affect the composition and structure of native plant, bird and soil arthropod communities, and displace endemic native species thereby promoting biological invasion by other alien species (León-Gamboa *et al.*, 2010; Pauchard *et al.*, 2015; Ziller *et al.*, 2005). *Pinus* spp. (pine) also have positive feedback with fire due to the accumulation of dry matter, this in turn results in greater intensity and frequency of fires (Cóbar-Carranza *et al.*, 2014; GISP, 2005; Paritsis *et al.*, 2018; Pauchard *et al.*, 2008, 2015; Raffaele *et al.*, 2016; Zalba *et al.*, 2008; **Chapter 1**, **Box 1.4**; **Chapter 3**, **sections 3.3.1.5.2** and **3.3.4.5**) and favours high *Pinus* spp. density post-fire (K. T. Taylor *et al.*, 2017). Other examples of invasive alien plant species with several records of negative impacts on different levels of ecological organization are *Prosopis juliflora* (mesquite; **Box 4.9**), *Impatiens glandulifera* (Himalayan balsam; e.g., Kiełtyk & Delimat, 2019), *Acacia longifolia* (golden wattle; e.g., Rascher *et al.*, 2011), *Cenchrus ciliaris* (buffel grass; e.g., Alves *et al.*, 2018; Bonney *et al.*, 2017), *Reynoutria japonica* (Japanese knotweed), *Robinia pseudoacacia* (black locust), and *Ailanthus altissima* (tree-of-heaven; e.g., Vilà *et al.*, 2010). Particularly, the latter three species are the invasive alien plants with the most widespread impacts across terrestrial habitats in European countries (Vilà *et al.*, 2010).

Box 4 9 ***Prosopis juliflora* (mesquite), an example of a high impact invasive alien plant.**

Prosopis juliflora (mesquite; **Figure 4.16**), a tree native to the Caribbean and tropical America, is considered one of the highest impact invasive alien trees (R. T. Shackleton *et al.*, 2014). It was deliberately introduced to 129 countries **(Figure 4.17)** to provide forage for livestock, for firewood, charcoal, as an ornamental, and to halt desertification and stabilize dunes in arid and semi-arid regions (Pasiecznik, 2001). However, this species has been documented to have negative impacts on native species, as well as on nature's contributions to people and good quality of life, throughout its introduced range (Patnaik *et al.*, 2017). Apart from its human-assisted spread, the species also spreads rapidly, aided by wild herbivores and livestock that feed on its pods and disperse its seeds. In the Afar region, Ethiopia, *Prosopis juliflora* is estimated to have invaded an area of about 1.17 million hectares (i.e., 12 per cent of the region) over a period of 35 years (Shiferaw, Schaffner, *et al.*, 2019).

Box 4 9

Figure 4 16 ***Prosopis juliflora* (mesquite) illustrations.**

Prosopis juliflora in the Cauvrey River delta in Tamil Nadu, India (top left), *Prosopis juliflora* wood being piled up for making charcoal (top right), bags of *Prosopis juliflora* charcoal being loaded up for transport to market (bottom left), and *Prosopis juliflora* flowers (bottom right). Photo credits: Bella S. Galil – CC BY 4.0 (top left) / Ankila J. Hiremath – CC BY 4.0 (top right, bottom left) / courtesy of Nirav Mehta – CC BY 4.0 (bottom right).

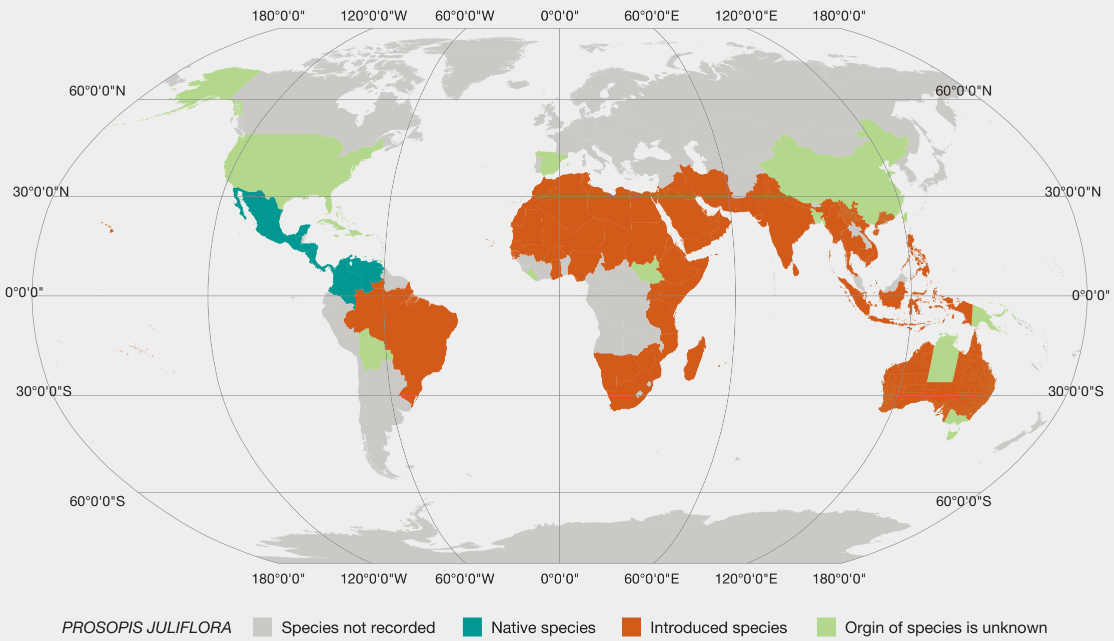


Figure 4 17 **Global distribution of *Prosopis juliflora* (mesquite).**

The invasive alien plant has been documented in many countries. Data source: Pasiecznik (2022).

Box 4 9

Impacts on nature

Negative impacts on nature mainly result from competition and habitat alteration. In the Brazilian Caatinga, *Prosopis juliflora* reduces the abundance of native species by more than 80 per cent, affecting seedling growth and mortality, and floristic composition, diversity, and structure of the native communities (Pegado *et al.*, 2006). It has had direct negative impacts on wildlife by altering natural grassland and wetland habitats (Mukherjee *et al.*, 2017; Sinha *et al.*, 2009).

Impacts on nature's contributions to people

One of the main negative impacts of *Prosopis juliflora* on nature's contributions to people, throughout its introduced range, is the loss of grazing lands, e.g., in East Africa (Bekele *et al.*, 2018; Mwangi & Swallow, 2008), and India (Duenn *et al.*, 2017; P. N. Joshi *et al.*, 2009). In Brazil and India, *Prosopis juliflora* has directly affected agriculture, competing with traditional short-cycle crops or encroaching tilled fields (Walter & Armstrong, 2014). It also affects agriculture indirectly due to increased crop raiding by wild herbivores as a result of reduction in wild forage availability (Sinha *et al.*, 2009). Furthermore, it has been shown that the water use of *Prosopis juliflora* impacts on water availability (Wise *et al.*, 2012). In the Afar Region, Ethiopia, the catchment water budget was estimated to be reduced by 3.1 to 3.3 billion m^3/year (Shiferaw *et al.*, 2021). The aggregated average social annual willingness to pay (**Glossary**) to manage the biological invasion in Afar, Ethiopia, and Baringo, Kenya, is estimated at US$6.1 million and US$4.2 million, respectively (Bekele *et al.*, 2018). *Prosopis juliflora* also provides benefits to people. For example, it is widely used by local communities in semi-arid regions of Brazil mainly for timber purposes (Guerra *et al.*, 2014), but also potentially as a natural pesticide and in the management of diseases in crop plants (Damasceno *et al.*, 2017). The fruits (pods) of *Prosopis juliflora* can be used to produce a number of food products; and they are extensively used for feeding livestock (Damasceno *et al.*, 2017; Duenn *et al.*, 2017). However, the livestock can only be fed up to a certain percentage by *Prosopis juliflora* pods, because exclusive feeding with these pods causes neurological disorders in the cattle (Patnaik *et al.*, 2017).

Impacts on good quality of life

Prosopis juliflora has more records of negative impacts than positive impacts on good quality of life. Reports from Africa demonstrate a negative effect of *Prosopis juliflora* on the occurrence of mosquito-borne human diseases. For example, in the Baringo area, Kenya, 40 to 60 per cent of local residents documented an increase in the incidence of malaria (Mwangi & Swallow, 2008). *Prosopis juliflora* flowers provide nectar for mosquito vectors of malaria, with higher numbers of female mosquitoes documented in Malian villages surrounded by *Prosopis juliflora* (Muller *et al.*, 2017). Further impacts include reduced access to grazing areas and water sources, resulting in conflicts among pastoralist communities due to resource scarcity; in India it has also been linked to conflicts between pastoralists and settled agriculturalists, due to livestock dispersing unwanted *Prosopis juliflora* into farmers' fields (Duenn *et al.*, 2017). In Ethiopia, reduction in grazing lands is leading to a breakdown of traditional customary laws as people seek new grazing areas disregarding the traditional users of these areas (Shiferaw, Bewket, *et al.*, 2019).

In heavily invaded areas, people have adapted to novel *Prosopis juliflora*-based livelihoods, especially making charcoal and harvesting the wood for sale; this livelihood diversification has enabled communities to cope better with losses of income from livestock or crops and respond to environmental shocks (Linders *et al.*, 2020; Sato, 2013; Walter & Armstrong, 2014). The longer term consequences of these adaptation processes seem to be context dependent: while studies in Africa found that utilization of the species was offsetting the losses, this was not expected to be sustainable in the future (Linders *et al.*, 2020; Wise *et al.*, 2012); whereas a study in India found that household incomes increased when the creation of small scale electricity generating facilities increased the demand for and prices of wood for energy generation following policy changes deregulating the electricity market (Sato, 2013).

Parts of the tree have traditionally been used for medicinal purposes, and people are adapting it for medicinal use in its introduced habitats (Damasceno *et al.*, 2017; Duenn *et al.*, 2017; Patnaik *et al.*, 2017).

Local extinctions caused by invasive alien species

A total of 105 alien species have been documented to have caused local extinctions of terrestrial native species, with a majority documented in tropical and subtropical dry and humid forests and Mediterranean forests, woodlands and scrub. Meanwhile, fewer local extinction caused by invasive alien species have been documented in tundra and high mountain habitats and urban/semi-urban habitats **(Table 4.7)**.

Invasive alien vertebrates are the main taxa responsible for local extinctions in terrestrial habitats (454 of 725 documented local extinctions have been caused by 36 invasive alien vertebrates). *Felis catus* (cat) has been documented as culpable in the greatest number of local extinctions, followed by *Vulpes vulpes* (red fox) and *Rattus Rattus* (black rat) **(Figure 4.18)**. These predatory invasive alien mammals have played a major role in the local extinction of native species in several terrestrial habitats (Doherty *et al.*, 2016; Radford *et al.*, 2018). Invasive alien invertebrates are the second taxa responsible for local extinctions of native species in terrestrial habitats (207 of 725 impacts with this magnitude have been caused by 33 invasive alien invertebrates), and most of these extinctions were registered on tropical and subtropical dry and humid forests, Mediterranean forests and temperate and boreal forests and woodlands. These invasive alien invertebrates include *Linepithema humile* (Argentine ant), *Solenopsis invicta* (red imported fire ant), *Anoplolepis gracilipes* (yellow crazy ant), and *Agrilus planipennis* (emerald ash borer). The emerald ash borer causes local extinctions of native plants through herbivory and is the focus of many studies because its larvae, feeding on ash trees, can kill the totality of ash varieties in tree stands, and, most recently, has been found to facilitate the spread of *Chionanthus* (fringetrees) in the northeast United States.

A similar number of invasive alien plants (31 species) have caused local extinctions of native species in terrestrial habitats. However, only 5 per cent (52 of 725 impacts) of all documented local extinctions have been caused by invasive alien plants. These invasive alien plant species include *Vachellia nilotica* (gum arabic tree), *Parthenium hysterophorus* (parthenium weed), and *Prosopis juliflora* (mesquite) that produced local extinctions of native species primarily due to competition (e.g., Duenn *et al.*, 2017) and poisoning or toxicity (e.g., Batish *et al.*, 2012).

In contrast, there are very few reports of local extinctions of native species (12 documented local extinctions) caused by only a few invasive alien microbes (5 invasive microbe species), which were documented on tropical and subtropical dry and humid forests and Mediterranean forests, woodlands and scrub. For example, the pathogenic *Batrachochytrium dendrobatidis* (chytrid fungus) has been associated with amphibian population declines, causing extinctions of frogs and salamanders in central and south America and Australia (Burrowes & De la Riva, 2017; Catenazzi *et al.*, 2011; Lampo *et al.*, 2008; Pounds *et al.*, 2006; Schloegel *et al.*, 2006).

Figure 4 18 **Examples of terrestrial invasive alien species which can cause local or global extinctions of native species.**

Felis catus (cat, top left), *Vulpes vulpes* (red fox, top right), *Rattus* spp. (rats, bottom left), *Boiga irregularis* (brown tree snake, bottom right). Photo credits: Mark Marathon, WM Commons – CC BY-SA 4.0 (top left) / Gregory "Slobirdr" Smith, flickr – CC BY-SA 2.0 (top right) / ngamanuimages – Copyright (bottom left) / U.S. Department of Agriculture, flickr – CC BY 2.0 (bottom right).

Table 4 7 **Main invasive alien species impacting nature in the terrestrial realm.**

List of alien species (top 10, by number of records of impacts) causing the maximum impacts on nature in the terrestrial realm, by the affected unit of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate: Microorganisms:

Units of Analysis	Taxa	Species	# Records
Temperate and boreal forests and woodlands		*Vulpes vulpes* (red fox)	31
		Linepithema humile (Argentine ant)	15
		Rattus spp. (rats)	12
		Lasius neglectus (invasive garden ant)	3
		Lymantria dispar (gypsy moth)	3
		Agrilus planipennis (emerald ash borer)	2
		Castor canadensis (North American beaver)	2
		Mustela erminea (ermine)	2
		Sciurus carolinensis (grey squirrel)	2
		Adelges piceae (balsam woolly adelgid)	1
Cultivated areas (incl. cropping, intensive livestock farming etc.)		*Anoplolepis gracilipes* (yellow crazy ant)	3
		Bombus terrestris (bumble bee)	3
		Pheidole megacephala (big-headed ant)	2
		Cenchrus ciliaris (buffel grass)	2
		Parthenium hysterophorus (parthenium weed)	2
		Paratrechina longicornis (longhorn crazy ant)	1
		Plagiolepis alluaudi (little yellow ant)	1
Deserts and xeric shrublands		*Vulpes vulpes* (red fox)	32
		Bromus spp. (bromegrasses)	3
		Bromus tectorum (downy brome)	2

Table 4 7

Units of Analysis	Taxa	Species	# Records
Deserts and xeric shrublands		*Linepithema humile* (Argentine ant)	1
		Cenchrus ciliaris (buffel grass)	1
Tropical and subtropical dry and humid forests		*Capra hircus* (goats)	31
		Anoplolepis gracilipes (yellow crazy ant)	24
		Boiga irregularis (brown tree snake)	14
		Pheidole megacephala (big-headed ant)	12
		Philornis downsi (avian vampire fly)	12
		Euglandina rosea (rosy predator snail)	10
		Wasmannia auropunctata (little fire ant)	10
		Vulpes vulpes (red fox)	9
		Sus scrofa (feral pig)	8
		Batrachochytrium dendrobatidis (chytrid fungus)	7
Temperate grasslands		*Cenchrus ciliaris* (buffel grass)	2
		Vulpes vulpes (red fox)	2
		Ageratina adenophora (Croftonweed)	1
		Bromus tectorum (downy brome)	1
		Panicum coloratum (klein grass)	1
		Rosa rugosa (rugosa rose)	1
		Bos taurus (cattle)	1
		Crocidura russula (greater white-toothed shrew)	1
Mediterranean forests, woodlands and scrub		*Vulpes vulpes* (red fox)	31
		Linepithema humile (Argentine ant)	29

Table 4 7

Units of Analysis	Taxa	Species	# Records
Mediterranean forests, woodlands and scrub		*Lasius neglectus* (invasive garden ant)	2
		Wasmannia auropunctata (little fire ant)	2
		Eucalyptus camaldulensis (red gum)	2
		Cydalima perspectalis (box tree moth)	1
		Pheidole megacephala (big-headed ant)	1
		Ceratocystis platani (canker stain of plane)	1
		Acacia saligna (coojong)	1
		Pinus radiata (radiata pine)	1
Tropical and subtropical savannas and grasslands		*Vulpes vulpes* (red fox)	15
		Vachellia nilotica (gum arabic tree)	7
		Wasmannia auropunctata (little fire ant)	5
		Anoplolepis gracilipes (yellow crazy ant)	3
		Canis lupus familiaris (dogs)	3
		Paratrechina fulva (tawny crazy ant)	2
		Solenopsis geminata (tropical fire ant)	2
		Capra hircus (goats)	2
		Columba livia (pigeons)	2
		Micropterus dolomieu (smallmouth bass)	2
Tundra and high mountain habitats		*Eucalyptus globulus* (Tasmanian blue gum)	1
		Vulpes vulpes (red fox)	1
Urban/Semi-urban		*Bombus terrestris* (bumble bee)	3
		Pheidole megacephala (big-headed ant)	3

Table 4 7

Units of Analysis	Taxa	Species	# Records
Urban/Semi-urban		*Linepithema humile* (Argentine ant)	2
		Parthenium hysterophorus (parthenium weed)	2
		Anoplolepis gracilipes (yellow crazy ant)	1
		Myrmica rubra (common red ant)	1
		Corvus splendens (house crow)	1

Positive impacts caused by invasive alien species on nature in the terrestrial realm

In the terrestrial realm, documented positive impacts on nature are mostly caused by invasive alien plants (**section 4.1.2**; **Box 4.3**). Highest numbers of invasive alien species causing positive impacts to native species can be found in temperate boreal forests and woodlands and temperate grasslands. Invasive alien plants causing the most documented positive impacts on native species are *Reynoutria japonica* (Japanese knotweed), *Impatiens glandulifera* (Himalayan balsam), and *Robinia pseudoacacia* (black locust). For instance, invaded areas by *Reynoutria japonica* showed higher abundances of bumblebees, overall insect diversity and hoverfly diversity than uninvaded areas (Davis *et al.*, 2018). Nonetheless, *Reynoutria japonica*, *Impatiens glandulifera*, and *Robinia pseudoacacia* are also invasive alien plant species with high numbers of negative impacts on nature on terrestrial habitats (**section 4.3.1**).

4.3.2.2 Patterns of negative and positive impacts of invasive alien species on nature in inland waters

In inland waters ecosystems the impacts of invasive alien species often act in synergy with other pressures, including unsustainable water abstraction, widespread habitat loss and degradation, overexploitation of natural resources, climate change, and other drivers of biodiversity change (Darwall *et al.*, 2018). However, in some cases, invasive alien species are the main driver contributing to native species extinctions and population declines; for example, the precipitous decline of critically endangered amphibians has been caused by the pathogenic *Batrachochytrium dendrobatidis* (chytrid fungus) (Dueñas *et al.*, 2021).

Concerns about inland waters ecosystems' vulnerability to invasive alien species have contributed to an increase in the number of studies on invasive alien species in inland waters (Ricciardi & Macisaac, 2011). Negative impacts of invasive alien species on nature in inland waters represent about 20 per cent of the total number of documented negative impacts caused by invasive alien species (2,113 of 10,822 impacts). A total of 230 invasive alien species have been documented to cause these impacts in inland waters.

Mechanisms and magnitude of impacts

Ecological impacts associated with invasive alien fishes include biotic homogenization (**Glossary**) and replacement of endemic species, spread of new diseases, changes in behaviour and diet shifts of native species (Gherardi, 2010; **Table 4.8**).

Impacts of invasive alien species on native inland waters biota and ecosystems are often synergistic and the result of multiple mechanisms such as predation and competition (Olden *et al.*, 2021), and complex interactions. For example, invasive alien freshwater mussels require fish hosts to complete their life cycle (Modesto *et al.*, 2018), and, in Sweden, declines in native crayfish species have been driven by the combined effects of hybridization, the transmission of crayfish plague and competitive exclusion with introduced crayfishes (Lodge *et al.*, 2012). Other complex interactions include the facilitation by some invasive alien species of the establishment of other invasive alien species (Simberloff & Von Holle, 1999), or the contribution of some invasive alien species to multiple stressors in their introduced habitat (M. C. Jackson *et al.*, 2016; Reynolds & Aldridge, 2021). There are synergistic interactions between invading species and cascading food-webs that may affect ecosystems within and beyond water bodies (Ricciardi & Macisaac, 2011). In North America, for example, the introduction of *Mysis*

Table 4 8 **Number of invasive alien species causing negative impacts on nature in inland waters.**

The number of invasive alien species (IAS) adversely impacting nature in the freshwater realm, and the number of documented negative impact by unit of analysis in relation to the maximum impact on native species: decline in performance, decline in population, local extinction or unspecified. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Unit of analysis	Maximum impact on native species							
	Decline in performance		Decline of population		Local extinction		Unspecified	
	IAS	Records	IAS	Records	IAS	Records	IAS	Records
Aquaculture areas	14	17	21	58	1	1	14	15
Wetlands	44	91	52	156	10	22	37	87
Inland surface waters and water bodies/ freshwater	132	613	125	636	54	168	94	249

relicta (opossum shrimp) to more than a hundred lakes, to stimulate the production of *Oncorhynchus nerka* (sockeye salmon), resulted in predation-driven declines of native zooplankton to such an extent that it led to the collapse of important planktivorous fish populations and to the decline of eagle and grizzly bear populations (C. N. Spencer *et al.*, 1991).

Decreases in native species' performance (34 per cent) or declines in local native populations (40 per cent) are the most commonly documented of all negative impacts on nature (2113 impacts) in inland waters. Local extinctions represent about 9 per cent of all documented impacts caused by invasive alien species in inland waters. For example, *Dikerogammarus villosus* (killer shrimp) caused the local extinction of the native amphipod *Gammarus duebeni* in Dutch water bodies through predation (Dick & Platvoet, 2000).

Impacted units of analysis

In inland waters, most documented impacts **(Table 4.8)** are from inland surface waters and water bodies (78.8 per cent), and fewer from wetlands (16.8 per cent) and areas used for aquaculture (4.3 per cent). Consequently, the documented number of invasive alien species causing impacts in the freshwater realm **(Table 4.8)** is larger for inland surface waters and water bodies (209) compared to wetlands (23) and areas used for aquaculture (28). Note that the same invasive alien species might be documented causing impacts in multiple units of analysis.

Invasive alien taxa most often documented causing negative impacts in inland waters

Some inland waters fish act as engineering species, profoundly affecting the environment. For example, *Cyprinus carpio* (common carp) and *Ctenopharyngodon idella* (grass carp) modify aquatic vegetation directly through uprooting or herbivory and indirectly through bioturbation and excretion, ultimately shifting the trophic status of water from clear to turbid (Matsuzaki *et al.*, 2009; Roberts *et al.*, 1995; Vilizzi *et al.*, 2015). *Salvelinus fontinalis* (brook trout), a widely introduced freshwater fish affecting food-webs and native diversity through various mechanisms (e.g., predation of various taxa including crustaceans, insects, amphibians and competition or hybridization with native fishes) is also causing high ecological impacts (Cucherousset *et al.*, 2007, 2008; Orizaola & Brana, 2006).

Despite not exceeding 2 per cent of total plant diversity, aquatic plants are vital in inland waters, shaping key processes such as primary production, oxygen release, and bank stabilization (Bolpagni, 2021). In Europe, more than half of the invasive alien species considered of concern according to the European Union Regulation 1143/2014, thus deemed highly impactful, are either aquatic or wetland plants. Dense mats of floating aquatic plants (e.g., *Pontederia crassipes* (water hyacinth)) can cause the complete cover of smaller water bodies and reduce the light available to submerged plants and phytoplankton, thus depleting dissolved oxygen and altering the composition of invertebrate communities (Hill *et al.*, 2020). Similar changes caused by thick underwater mats of *Myriophyllum spicatum* (spiked watermilfoil), leading to the decline of native macrophytes and invertebrates, have been observed in water bodies of North America (Boylen *et al.*, 1999; Kauffman *et al.*, 2018; S. J. Wilson & Ricciardi, 2009). Furthermore, *Myriophyllum spicatum* can affect native North American milfoils through hybridization; with the hybrid watermilfoil *Myriophyllum spicatum* x *Myriophyllum sibiricum* exhibiting an increase in reproductive potential and surface cover compared to its parental taxa (Glisson & Larkin, 2021).

The crayfish group causes many negative impacts in inland waters habitats. For instance, *Faxonius limosus* (spiny-cheek crayfish) and *Procambarus clarkii* (red swamp crayfish) interfere with water quality regulation, habitat maintenance and nutrient cycling through their burrowing activities, and decrease the abundance of macrophytes by feeding and stalk-cutting, reducing the availability of refuges for other species (Lodge *et al.*, 2012). In Portugal, the invasive *Pacifastacus leniusculus* (American signal crayfish) threatens the survival of *Margaritifera margaritifera* (freshwater pearl mussel), a critically endangered species in Europe, through predation (R. Sousa *et al.*, 2019). The translocation of live crayfish for aquaculture purposes has also facilitated the transmission of diseases that are potentially lethal to native crayfish (e.g., *Aphanomyces astaci* (crayfish plague); Martín-Torrijos *et al.*, 2018) and of ectosymbiotic branchiobdellidans (e.g., *Xironogiton victoriensis* carried by its host, *Pacifastacus leniusculus*; Gelder & Williams, 2015). This creates opportunities for novel associations between, for example, alien branchiobdellidans and native crayfish, *Xironogiton victoriensis* and the endangered native *Austropotamobius pallipes* (Atlantic stream crayfish) in Spain (Martín-Torrijos *et al.*, 2018), whose consequences are difficult to predict.

Python bivittatus (Burmese python) is another emerging invasive alien species, established in southern Florida. Free-ranging *Python bivittatus* have consumed a wide variety of birds, mammals, and one reptile, the *Alligator mississippiensis* (American alligator; Dove *et al.*, 2011; Guzy *et al.*, 2023; Rochford *et al.*, 2010; Snow *et al.*, 2007). Large species of mammals and birds are also vulnerable to predation by invasive pythons; *Lynx rufus* (bobcat), *Odocoileus virginianus* (white-tailed deer), and *Mycteria americana* (wood stork) have been consumed by *Python bivittatus* in the Everglades National Park (Dove *et al.*, 2011; Guzy *et al.*, 2023; Rochford *et al.*, 2010; Snow *et al.*, 2007). This large and voracious predator is directly responsible for the severe decline of several mammal populations (e.g., raccoons, opossums and rabbits; McCleery *et al.*, 2015). By reducing populations of their native predators, invasive alien pythons might have a potential indirect positive impact on non-prey species, for example by decreasing nest predation on native turtles (Willson, 2017).

Local extinctions caused by invasive alien species in the inland waters realm

Several invasive alien invertebrates cause local extinctions, including *Dreissena polymorpha* (zebra mussel) and *Faxonius limosus* (spiny-cheek crayfish) (**Figure 4.19**; **Table 4.9**). The introduction of invasive alien fishes, such as *Salvelinus fontinalis* (brook trout; Cucherousset *et al.*, 2007, 2008; Orizaola & Brana, 2006) and *Oreochromis niloticus* (Nile tilapia; Angienda *et al.*, 2011; Wise *et al.*, 2007), has caused local extinctions of native fishes and amphibians in all units of analysis of the inland waters realm (Cucherousset & Olden, 2011; Ellender & Weyl, 2014; **Table 4.9**). The best cited example of predation-induced extinction is the local extinction of about 200 species of endemic cichlid fish following the introduction of *Lates niloticus* (Nile perch) in Lake Victoria (Witte *et al.*, 1992; **Box 4.10**). *Clarias gariepinus* (North African catfish) has also been documented as causing local extinctions in areas used for aquaculture purposes. In India, *Clarias gariepinus* is considered responsible for the decrease of vertebrate species richness from several ponds during the post-monsoon season (Gopi & Radhakrishnan, 2002).

Plants such as *Pontederia crassipes* (water hyacinth) and *Pistia stratiotes* (water lettuce), have also caused local extinctions, mostly in wetlands **(Table 4.9)**. *Pistia stratiotes* causes changes in physiochemical properties of invaded water bodies, affecting water quality and altering macrophyte communities leading, in some cases, to the local extinction of native species such as several species of the pondweed *Potamogeton* in Slovenia (Jaklič *et al.*, 2020).

Inland waters molluscs represent one of the most diverse, but also highly threatened groups, in the inland realm (Böhm *et al.*, 2021). The diversity and the functions they provide (e.g., biofiltration, nutrient cycling and storage, substrate and trophic resources) are essential to aquatic ecosystems and susceptible to changes (Vaughn, 2018). Invasive alien molluscs can cause the decline of phytoplankton biomass or native mussels abundance. For instance, *Dreissena polymorpha* (zebra mussel) is responsible for the 10-fold increase in the rate of local extinction of native mussels in the Great Lakes region (Ricciardi *et al.*, 1998). *Pomacea canaliculata* (golden apple snail) is another example of an invasive alien mollusc responsible for the increase of phytoplankton biomass through the release of nutrients when grazing (Strayer, 2010), and outcompeting native apple snails in Indonesia (Marwoto *et al.*, 2020).

Figure 4 19 **Examples of inland waters invasive alien species causing local/global extinctions of native species.**

Pontederia crassipes (water hyacinth, top left), *Salvelinus fontinalis* (brook trout, top right), *Dreissena polymorpha* (zebra mussel, bottom left), *Pacifastacus leniusculus* (American signal crayfish, bottom right). Photo credits: Philip, Adobe Stock – Copyright (top left) / slowmotiongli, Adobe Stock – Copyright (top right) / Thirdwavephoto, WM Commons - CC BY 4.0 (bottom left) / LFRabanedo, Shutterstock – Copyright (bottom right).

Table 4 9 **Examples of invasive alien species causing local extinctions in inland waters, by the affected unit of analysis.**

The list of invasive alien species (top 10, by number of records of impacts) causing local extinctions on nature in inland waters, by the affected unit of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate:

Unit of analysis	Taxa	Invasive alien species	#records of local extinctions
Aquaculture areas		*Clarias gariepinus* (North African catfish)	1
Wetlands		*Python bivittatus* (Burmese python)	5
		Sporobolus densiflorus (denseflower cordgrass)	3
		Pomacea canaliculata (golden apple snail)	1
		Raffaelea lauricola (laurel wilt)	1
		Sporobolus alterniflorus (smooth cordgrass)	1

Table 4 9

Unit of analysis	Taxa	Invasive alien species	#records of local extinctions
Wetlands		*Typha angustifolia* (lesser bulrush)	1
		Typha ×glauca (hybrid cattail)	1
		Oreochromis spp. (tilapia)	1
Inland surface waters and water bodies/freshwater		*Pontederia crassipes* (water hyacinth)	17
		Salvelinus fontinalis (brook trout)	9
		Dreissena polymorpha (zebra mussel)	8
		Pomacea canaliculata (golden apple snail)	8
		Pistia stratiotes (water lettuce)	8
		Lates niloticus (Nile perch)	8
		Oreochromis niloticus (Nile tilapia)	8
		Faxonius limosus (spiny-cheek crayfish)	7
		Procambarus clarkii (red swamp crayfish)	7
		Pacifastacus leniusculus (American signal crayfish)	6

Conflict species causing both positive and negative impacts

Some invasive alien species can be referred to as conflict species (**Chapter 1**, **section 1.5.2**; **Chapter 5**, **section 5.6.1.2**), causing both positive and negative impacts, although this should be interpreted with caution as it is context-dependent **(Box 4.10)**. Such species are challenging to manage, as they affect stakeholders in different ways (**Chapter 5**, **section 5.6.1.2**). Examples of conflict species include invasive alien macrophytes providing refuge from predators to various native species or limiting bank erosion. Likewise, invasive alien crayfish provide food or shelter for other native species, are adequate for human consumption and can be appreciated for their aesthetic properties and cultural or spiritual values (Emery-Butcher *et al.*, 2020; Vaughn, 2018; **section 4.4.1**). Many crayfish species, like the North American *Faxonius immunis* (calico crayfish) and the parthenogenetic form of *Procambarus fallax* (slough crayfish), are kept as ornamental species in aquaria and ponds throughout Europe. This has led to a flourishing pet trade and to the inevitable escape or introduction of the crayfish in the wild with negative impacts on the native fauna (Faulkes, 2010; Holdich *et al.*, 2009; Martin *et al.*, 2010; Nonnis Marzano, 2009; **Chapter 3**, **section 3.2.3.2**) and on good quality of life, including cultural, social and ethical values, in many countries (Gherardi, 2011; Swahn, 2004).

Box 4 10 Fishes as examples of invasive alien species with both positive and negative impacts.

Invasive alien species may cause both positive and negative impacts on nature, nature's contributions to people and good quality of life (Zengeya *et al.*, 2017). Many fish have been intentionally introduced to enhance fisheries or as control agents, providing remarkable cautionary examples.

Lates niloticus (Nile perch), introduced in Lake Victoria, East Africa, to enhance the fishery, is a prime example (Balirwa *et al.*, 2003; **Figure 4.20**). Lake Victoria's fish fauna was comprised of about 500 endemic haplochromine cichlid species, two tilapiine species and 46 other species belonging to 12 families

Box 4 10

(Witte *et al.*, 2013). As increasing fishing pressure reduced the native tilapiine cichlids and other large fish species' populations, *Lates niloticus* and four tilapiine cichlids were introduced into the lake in the 1950s (Aloo *et al.*, 2017; Gichuru *et al.*, 2018; Luomba, 2016; Marshall, 2018). *Lates niloticus* biomass peaked at around 2.3 million tonnes in 1999, and accounted for 92 per cent of total fish biomass but fell to less than 300,000 tonnes in 2008, with average length declining from 51.7 cm to 26.6 cm, significantly below the required minimum size of 50 cm for export (Talma *et al.*, 2014). Dramatic changes ensued: *Lates niloticus* and *Oreochromis niloticus* (Nile tilapia) increased, as well as eutrophication of the lake, and the wetlands declined. The haplochromine cichlids were the most severely hit, with most species presumed extinct. These introductions were an economic success: the annual catch is estimated at US$544 million locally, in addition to US$243 million in exports in 2003 (Balirwa, 2017), at the price of the greatest documented extinction of vertebrates (Kaufman, 1992), with an estimated loss of 200 endemic fish species (Witte *et al.*, 1992).

Figure 4 20 ***Lates niloticus* (Nile perch).**

Photo credit: Fotogien, Shutterstock – Copyright.

The widely introduced *Oreochromis niloticus* and four species of carps – *Ctenopharyngodon idella* (grass carp), *Hypophthalmichthys molitrix* (silver carp; **Figure 4.21**), *Hypophthalmichthys nobilis* (bighead carp), and *Cyprinus carpio* (common carp), account for more than a third of the global freshwater fish production and contribute to global food security (FAO, 2020). These fish are listed among the world's worst invasive alien species (Lowe *et al.*, 2000). *Oreochromis niloticus* threatens native tilapia in Africa through hybridization and competition (Canonico *et al.*, 2005). *Cyprinus carpio* suspends sediments, increasing nutrient availability and turbidity, suppressing macrophyte growth (Vilizzi *et al.*, 2015). *Ctenopharyngodon idella* modifies aquatic vegetation through uprooting or herbivory and has transmitted parasites which threaten wild fish (Cucherousset & Olden, 2011).

Lakes and rivers worldwide were stocked with salmonids, including *Oncorhynchus mykiss* (rainbow trout), *Salmo trutta* (brown trout), and *Salvelinus fontinalis* (brook trout), for commercial and recreational exploitation. These top predators brought about profound ecological changes: predation on native fauna can reduce amphibian and reptile populations, led to changes in zooplankton and benthic macroinvertebrate species composition and size structure, alteration of nutrient cycling, competition for food and habitat, hybridization with native trout species, and disease transmission (Krueger & May, 1991; P. Jones & Closs, 2018; Miró & Ventura, 2013). Management of these and other conflict species depends on better balancing of competing goals and perspectives (Vigliano & Alonso, 2007; Ellender *et al.*, 2014; Zengeya *et al.*, 2017; Beever *et al.*, 2019).

Box 4 10

Figure 4 21 ***Hypophthalmichthys molitrix* (silver carp).**

Photo credit: Ryan Hagerty/USFWS, flickr – Public domain.

4.3.2.3 Patterns of negative and positive impacts of invasive alien species on nature in the marine realm

The database of impacts developed through this chapter includes about 900 articles (2,350 reported impacts) documenting quantitative observational/experimental studies of impacts of invasive alien species in the marine realm. There are 159 documented invasive alien marine species causing 1,414 negative impacts on nature, and 72 invasive alien species causing 566 positive impacts. Some of the impacts could not be given a direction (**section 4.1.2**), for example impacts on abiotic ecosystem changes.

Impacted units of analysis

Most impacts of invasive alien species in the marine realm have been documented in shelf ecosystems (i.e., the shallow seafloor, between the shoreline and the shelf break, generally less than 200m in water depth; **Table 4.10**). The complex interactions among invasive alien populations and the host ecosystems (**Chapter 1**, **section 1.5**; **Boxes 4.3** and **4.5**), the functions they most often affect, the relationships between changes to ecosystems, communities, and populations, and the long-term responses of ecosystems to interactions with multiple anthropogenic activities, appear to offer insurmountable challenges in the marine realm, limiting the ability to assess the overall impact of invasive alien species on marine ecosystems (Fulton *et al.*, 2003).

Mechanisms of impacts

Marine invasive alien species have been shown to have differential impacts on native taxa within a biome, among different regions and ecosystems, from local extinction to food provision to rare and endangered species **(Box 4.11)**.

Table 4 10 **Number of impacts caused by invasive alien species in the marine realm.**

a. Number of invasive alien species documented as causing negative impacts on nature in the marine realm, by the affected unit of analysis, b. Number of records of negative impacts on nature in the marine realm, by the affected unit of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

a.

	Decline in performance	Decline in population	Local extinction	Unspecified
Shelf	69	116	59	46
Ocean	0	0	0	0

b.

	Decline in performance	Decline in population	Local extinction	Unspecified
Shelf	246	794	278	99
Ocean	0	0	0	0

Box 4 11 ***Magallana gigas*** **(Pacific oyster) in European Seas.**

Magallana gigas (Pacific oyster; **Figure 4.22**) is the most widely cultivated and harvested shellfish species in Europe, with production totalling 142,000 tons, valued at 295 million euros (US$304 million) in 2007 (Miossec *et al.*, 2009). It is also a highly invasive ecosystem engineer, forming reefs on hard and soft bottoms, effecting large structural changes in littoral communities. In the Wadden Sea, *Magallana gigas* brought about a shift in dominance from mussels to oysters which entailed changes of associated organisms (Kochmann *et al.*, 2008). Yet, these complex structures provide habitat heterogeneity that can result in increased species richness, abundance, biomass, and diversity, and in the case of the Wadden Sea, replacing the ecological function of the native *Mytilus edulis* (common blue mussel) (Markert *et al.*, 2010). A field experiment revealed that epibenthic faunal abundance and biomass was higher on (dead) oyster shells than on live animals, both favouring fish and larger invertebrate species, likely to retain the changes to benthic community structure even in the case of mass mortalities (Norling *et al.*, 2015).

Figure 4 22 ***Magallana gigas*** **(Pacific oyster) reef, Sylt I., Germany, Wadden Sea.**

Photo credit: G. Nehls – CC BY 4.0.

Box 4 11

In the Bay of Mont-St.-Michel, France, extensive *Sabellaria alveolata* (honeycomb worm) reefs were damaged through trophic competition, increased silt deposition, and recreational oyster harvesting leading to trampling, breakage, and reef fragmentation (Desroy *et al.*, 2011). The proliferating *Magallana gigas* beds impacted on birds as well: Waser *et al.* (2016) found that the abundances of four bird species in the Dutch Wadden Sea, *Larus canus* (common gull), *Somateria mollissima* (common eider), *Haematopus ostralegus* (Eurasian oystercatcher), and *Calidris canutus* (red knot) were reduced where mussel beds were replaced with oyster beds, which the birds were unable to feed on. Herbert *et al.* (2018) noted that in southeast England, areas colonized by oysters were utilized by greater numbers of oystercatchers and *Numenius* spp. (curlews), but smaller numbers of smaller shorebirds. *Larus argentatus* (European herring gull), too were disadvantaged by the replacement of mussel beds with oyster beds (Markert *et al.*, 2013). Yet, *Larus argentatus* has soon adapted and adopted a shell-dropping behaviour utilizing pavements and parking lots (Cadée, 2001).

Magallana gigas have served as a major vector for introduction of algae, invertebrates and pathogens (Mineur *et al.*, 2007; Wolff & Reise, 2002). Mineur *et al.* (2014) list 48 species that have likely been introduced through the Pacific Northwest to Europe route, along with the oyster trade, including notorious invasive alien species such as *Codium fragile* (dead man's fingers), *Sargassum muticum* (wire weed), *Undaria pinnatifida* (Asian kelp), the sea squirts *Botrylloides violaceus* (violet tunicate), *Didemnum vexillum* (carpet seas quirt), and *Styela clava* (Asian tunicate). The intrahemocytic parasite *Bonamia ostreae*, protozoan parasite *Marteilia refringens*, the *Ostreid herpesvirus* (OsHV-1), and the two species of parasitic copepods *Mytilicola orientalis* (oyster redworm) and *Myicola ostreae* have all caused massive mortalities. Mineur *et al.* (2014) lay out a compelling case that the periodic disease outbreaks, affecting farmed *Magallana gigas* in Europe and causing major production disruptions and losses, originate in the massive imports of stock. Although providing relief to the industry in the immediate term, the translocations invariably introduce new disease agents.

Local extinctions caused by invasive alien species in the marine realm

Although the number of quantitative observational and experimental impact studies is limited, and most studies focus on sessile biota, shallow water and economically important species, marine invasive alien species have been documented as having significant impacts and causing local extinctions (**Table 4.11**; **Figure 4.23**). *Pterois volitans* (red lionfish) and *Caulerpa taxifolia* (killer algae) are listed among the top 10 invasive alien species that have been documented as causing most local extinctions globally (but see Albins, 2015; Bacheler *et al.*, 2022; Ballew *et al.*, 2016; Ingeman, 2016; Verlaque & Fritayre, 1994; **Table 4.11**).

In the marine realm, most documented local extinctions occur on the shallow shelf, and eight of the 10 invasive alien species causing them belong to the sessile biota: in descending order, *Caulerpa taxifolia* (killer algae), *Mytilus galloprovincialis* (Mediterranean mussel), *Caulerpa cylindracea* (green algae), *Pyura praeputialis* (cunjuvoi), *Halophila stipulacea* (halophila seagrass), *Womersleyella setacea* (red alga), *Carijoa riisei* (branched pipe coral), *Kappaphycus alvarezii* (elkhorn sea moss).

The sole exception in **Table 4.11** is *Pterois volitans* (red lionfish), a voracious piscivore denuding the vestigial reefs in the tropical west Atlantic, documented as causing significant reduction in density, biomass and species richness of small native reef fish (Albins, 2015).

Globally, in the marine realm, documented local extinctions through biofouling have been mostly caused by *Kappaphycus alvarezii* (elkhorn sea moss), *Carijoa riisei* (branched pipe coral), *Mytilopsis sallei* (Caribbean false mussel), *Polydora websteri* (mud blister worm), *Pyura praeputialis* (cunjuvoi), *Ciona intestinalis* (sea vase), *Didemnum* spp. (colonial tunicates), *Mytella strigata* (Charru mussel), and *Mytilus galloprovincialis* (Mediterranean mussel). Documented local extinctions through competition have been mostly caused by *Caulerpa cylindracea* (green algae), *Mytilus galloprovincialis* (Mediterranean mussel) and *Caulerpa taxifolia* (killer algae). Documented local extinctions through ecosystem change have been mostly caused by *Caulerpa cylindracea* (green algae), *Mytilus galloprovincialis* (Mediterranean mussel), *Pyura praeputialis* (cunjuvoi), *Eucheuma denticulatum* (eucheuma seaweed), *Womersleyella setacea* (red alga) and *Crepidula fornicata* (American slipper limpet). Documented local extinctions through herbivory have been mostly caused by *Carcinus maenas* (European shore crab), *Siganus* spp. (rabbitfish) and *Littorina littorea* (common periwinkle). Documented local extinctions through parasitism have been mostly caused by *Anguillicola crassus* (eel swimbladder nematode), *Haplosporidium nelsoni* (MSX oyster pathogen) and *Loxothylacus panopaei* (parasitic barnacle). Finally, documented local extinctions through toxicity have been mostly caused by *Caulerpa taxifolia* (killer algae) **(Figure 4.23)**.

Main invasive alien species causing negative impacts in the marine realm

Anguillicola crassus (eel swimbladder nematode), a blood-feeding swimbladder parasitic nematode in eels, native to

Table 4 11 **Example of invasive alien species causing local extinctions in the marine realm.**

The list of invasive alien species causing local extinctions on nature in the marine realm. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate:

Taxa	Species	Records number
Vertebrate	*Pterois volitans* (red lionfish)	39
Plants	*Caulerpa taxifolia* (killer algae)	21
Invertebrate	*Mytilus galloprovincialis* (Mediterranean mussel)	20
Plants	*Caulerpa cylindracea* (green algae)	17
Invertebrate	*Pyura praeputialis* (cunjuvoi)	11
Invertebrate	*Carcinus maenas* (European shore crab)	9
Plants	*Halophila stipulacea* (halophila seagrass)	8
Plants	*Womersleyella setacea* (red alga)	8
Invertebrate	*Carijoa riisei* (branched pipe coral)	7
Plants	*Kappaphycus alvarezii* (elkhorn sea moss)	2

eastern Asia, has been widely introduced with its native host *Anguilla japonica* (Japanese eel) for stocking and farming in Europe and North America. It is considered to have contributed to the collapse of the *Anguilla anguilla* (European eel) population. The parasite reduces endurance, while damage to the swimbladder impairs buoyancy control. High infection levels can reduce swimming performance, likely rendering the eels more susceptible to potting, predation, and hindering them from reaching their spawning grounds (Newbold *et al.*, 2015; Palstra *et al.*, 2007; Sjöberg *et al.*, 2009; Sprengel & Luchtenberg, 1991). Mass mortalities of wild eels infected with *Anguillicola crassus* in Lake Balaton, Hungary, as well as laboratory results, suggested that infected eels may have been more stressed than uninfected eels by the reduced oxygen levels under high water temperatures or increased concentrations of toxicants (Bálint *et al.*, 1997; Molnár, 1993; Molnár *et al.*, 1991).

Carcinus maenas (European shore crab), native to European and North African coasts, has invaded both coasts of North America, south-eastern America, southern Australia and South Africa. It has contributed to decline in native soft-shell clams, *Mya arenaria* (sand gaper), off north-eastern America, reducing its density, and inducing deeper burrowing (de Rivera *et al.*, 2011; Floyd & Williams, 2004; Whitlow, 2010). Mortality of small *Crassostrea virginica* (eastern oyster) was significantly higher in the presence of *Carcinus maenas* (Poirier *et al.*, 2017). Off central California, *Carcinus maenas* reduced the abundance of the native *Hemigrapsus oregonensis* (yellow shore crab), markedly decreased its body size and caused it to shift its habitat to the high intertidal zone (de Rivera *et al.*, 2011; Grosholz *et al.*, 2000). *Carcinus maenas*' predation on the Tasmanian *Katelysia scalarina* (sand cockle) reduced its population (Walton *et al.*, 2002). *Chondrus crispus* (carrageen), a unique strain of the red alga, found solely amongst clumps of *Mytilus edulis* (common blue mussel) in a coastal lagoon in Atlantic Canada, was wiped out coinciding with *Carcinus maenas* preying on the mussel (Yorio *et al.*, 2020). *Carcinus maenas* accounted for steep declines in in faunal organisms (Gregory & Quijón, 2011). *Zostera marina* (eelgrass) beds have been declining as a result of uprooting, grazing and cutting by *Carcinus maenas* (Garbary *et al.*, 2004; B. R. Howard *et al.*, 2019; Malyshev & Quijón, 2011; Matheson *et al.*, 2016).

Carijoa riisei (branched pipe coral), native to the Indo-Pacific, has spread to Hawaii and the western tropical Atlantic (Concepcion *et al.*, 2010; Grigg, 2003; Kahng & Grigg, 2005; Sánchez & Ballesteros, 2014). A large-scale survey (200 km^2) of Maui's black corals revealed that at depths between 75 and 110 m up to 90 per cent of the colonies of *Antipathes dichotoma* (black coral) and *Antipathes grandis* (Pine coral) are dead, having been overgrown by *Carijoa*

Figure 4 23 **Examples of marine invasive alien species causing local extinctions of native species.**

Pterois volitans (red lionfish, top left), *Caulerpa* sp. (top right), *Mytilus galloprovincialis* (Mediterranean mussel, bottom left), *Carcinus maenas* (European shore crab, bottom right). Photo credits: plus69, Adobe Stock – Copyright (top left) / Coughdrop12, WM Commons – CC BY-SA 4.0 (top right) / Peter Southwood, WM Commons – CC BY-SA 4.0 (bottom left) / Nicolás Battini – CC BY 4.0 (bottom right).

riisei (Grigg, 2003). It also fouls *Myriopathes* spp. (feathery black corals) and *Leptoseris* spp. (scleractinian plate corals) (Kahng, 2007; Kahng & Grigg, 2005). In the tropical eastern Pacific, *Carijoa riisei* overgrew *Pacifigorgia* seafans and *Leptogorgia* seawhips, caused community-wide octocoral mortalities, and the local extinction of some *Muricea* spp. (seafans; Sánchez & Ballesteros, 2014).

Caulerpa cylindracea (green algae), native to Australia, was first documented in the Mediterranean in the early 1990s, where it soon spread throughout the sea, forming dense meadows. The alga modifies habitat structure in terms of repartition of the available substrate (i.e., enhancing sediment accumulation, favours algal turfs over erect algal forms and enables them to monopolize space) (Bulleri *et al.*, 2010). Such changes affect the associated invertebrate assemblages, algae-native species richness, cover and diversity decreased (Baldacconi & Corriero, 2009; Bulleri & Piazzi, 2015; Klein & Verlaque, 2009; Piazzi, Balata, & Cinelli, 2007; Piazzi, Balata, Foresi, *et al.*, 2007; Piazzi *et al.*, 2001; Vázquez-Luis *et al.*, 2008). The effects of the colonization persist after the removal of the alga and the recovery of the assemblages appears to be quite slow: species numbers, total cover and erect perennial species cover were significantly lower than in the non-invaded plots 18 months after removal and exclusion of *Caulerpa cylindracea* (Klein & Verlaque, 2011; Piazzi & Ceccherelli, 2006).

Caulerpa taxifolia (killer algae) is a green alga native to tropical Australia. Since the 1980s, a cold-resistant clone has become notorious for high profile invasions in the Mediterranean, and in California, United States and Australia in the 2000s. The mat-forming invasive form of *Caulerpa taxifolia* grows rapidly, smothers seagrass beds and other benthos, replacing native macroalgal and seagrass communities. It causes a decrease in number, width, longevity of leaves, chlorosis and necrosis, and finally death of shoots of the native *Posidonia oceanica* (Neptune grass) in the Mediterranean. Furthermore, seagrass beds have never recovered their initial density, even after the decrease in *Caulerpa taxifolia* (de Villèle & Verlaque, 1995; Dumay *et al.*, 2002; Molenaar *et al.*, 2009). In Australia, canopy covers of *Posidonia australis* (fibreball weed) and *Zostera capricorni* (garweed) were significantly reduced (Glasby, 2013). Invertebrate assemblages (e.g., Anomura, Peracarida, Decapoda, Echinoidea, Bivalvia and Gastropoda) declined (Francour *et al.*, 2009), but *Caulerpa taxifolia* promotes an overall increase in nematode species richness by favouring species that were absent from the native environments (Gallucci *et al.*, 2012). The density of fish such as the commercially important *Mullus surmuletus* (red mullet) has declined, compared to native seagrass meadows (Harmelin-Vivien *et al.*, 1996; Levi & Francour, 2004).

Cercopagis pengoi (fishhook waterflea), a planktonic cladoceran crustacean native to the Ponto-Aralo-Caspian Basin, has spread to the Baltic Sea. It is a voracious predator that markedly reduces the density of its prey (cladocerans, copepods) (Lehtiniemi & Gorokhova, 2008; Ojaveer *et al.*, 2004; Põllumäe & Kotta, 2007). The population of the native cladoceran *Bosmina (Eubosmina) coregoni* (large long-nosed waterflea) has significantly declined after the invasion (Kotta *et al.*, 2006). The reduction in zooplankton abundance may result in higher concentrations of phytoplankton (owing to reduced grazing by zooplankton), and may ultimately aggravate problems of eutrophication. Yet, *Cercopagis pengoi* has become an important food item for the three-spined and nine-spined sticklebacks, herring, sprat, and smelt (Antsulevich & Välipakka, 2000; Gorokhova *et al.*, 2004; Ojaveer *et al.*, 2004; Ojaveer & Lumberg, 1995).

Crepidula fornicata (American slipper limpet), native to the Atlantic coast of North America, has unintentionally been introduced to the Pacific coast as well as to Europe with American oysters, and has spread throughout the Atlantic coast. *Crepidula fornicata* reduces growth and increases mortality of fouled commercially important *Mytilus edulis* (common blue mussel; Thieltges, 2005a; Thieltges & Buschbaum, 2007). Yet, it reduces the infection success of cercariae and thus their parasite load (Thieltges *et al.*, 2009), and reduces *Asterias rubens* (common starfish) predation (Thieltges, 2005b). Dense reef-like populations fundamentally alter the physical and chemical composition of the sediment when forming a novel substrate for sessile invertebrates (i.e., ascidians, tubicolous worms, bivalves) and shelters vagile invertebrates, at the loss of the infauna, deterred by the putrid biodeposits (Blanchard, 2009). Even at a moderate presence of *Crepidula fornicata*, species composition differs from the composition in its absence (de Montaudouin & Sauriau, 1999; Vallet *et al.*, 2001). Accumulated shell debris also reduces suitable habitat for commercially valuable native flatfish (Kostecki *et al.*, 2011; Le Pape *et al.*, 2004).

Didemnum vexillum (carpet sea squirt), native to Japan, is a colonial tunicate species, widely introduced in temperate cold seas. Its massive encrusting mats, over-growing sessile biota, natural and man-made hard substrates, outcompetes other tunicates, hydroids, seaweeds, sponges, bivalves, and reduces areas suitable for settlement (Bullard *et al.*, 2007; Lengyel, 2009; Valentine, Carman, *et al.*, 2007; Valentine, Collie, *et al.*, 2007). Fouled mussels and oysters have decreased growth rates and lower condition index; the swimming ability of fouled *Placopecten magellanicus* (Atlantic deep-sea scallop) is reduced, limiting their ability to escape predation and access food-rich habitats, thus affecting their survival (Dijkstra & Nolan, 2011; Kaplan *et al.*, 2017). *Didemnum vexillum* fouling result in economic losses due to direct impact on biomass of farmed species, equipment and trade restrictions (Fletcher *et al.*, 2013).

Eucheuma denticulatum (eucheuma seaweed), a red alga native to the tropical western Pacific, has been widely introduced for cultivation as one of the primary sources of carrageenan. *Eucheuma denticulatum* spread from farms into the surrounding ecosystems, overgrows and outcompetes reef-building corals, reduces seagrass beds, macroalgae, abundance and biomass of macrofauna, as well as on the benthic microbial processes and meiofauna populations. These modifications are apparent in the significant difference in the catch composition, trophic groups and diet of fish collected on coral, seagrass, sand and seaweed farms (Eggertsen *et al.*, 2021; Eklöf *et al.*, 2005, 2006; Johnstone & Olafsson, 1995; Kelly *et al.*, 2020; Ólafsson *et al.*, 1995; Tano *et al.*, 2015; Yahya *et al.*, 2020).

Halophila stipulacea (halophila seagrass), native to the Red Sea, Persian Gulf and Indian Ocean, has spread to the Mediterranean and Caribbean seas, where it forms extensive monospecific mat-forming meadows. It has displaced the native seagrasses *Syringodium filiforme* (manatee grass), *Halodule wrightii* (shoalweed), and *Halophila decipiens* (Caribbean seagrass) off Dominica, Lesser Antilles, and *Thalassia testudinum* (turtle grass) in Bonaire (Muthukrishnan *et al.*, 2020; Smulders *et al.*, 2017; Steiner & Willette, 2013, 2015). Continued invasion and subsequent loss of native seagrasses reduce key juvenile fish habitats in the Virgin Islands, United States (Olinger *et al.*, 2017). Fish, as well the native sea urchin, *Tripneustes ventricosus* (white urchin), were twice as abundant in meadows of *Thalassia testudinum* as in *Halophila stipulacea* in Bonaire and the Grenadines, respectively (Becking *et al.*, 2014; Scheibling *et al.*, 2018). Similarly, in the Mediterranean, *Halophila stipulacea* displaced the native *Cymodocea nodosa* (slender seagrass; Sghaier *et al.*, 2014), and the epiphytic assemblages on the latter were more abundant and more diversified (Mabrouk *et al.*, 2021).

Kappaphycus alvarezii (elkhorn sea moss), a red alga native to Southeast Asia, has been widely introduced for cultivation as one of the primary sources of carrageenan. In the Gulf of Manaar, India, it has been documented as shadowing and smothering corals, seagrass, sponges and thus affecting the diverse reef-associated fauna (Chandrasekaran *et al.*, 2008; Kamalakannan *et al.*, 2010, 2014; Patterson *et al.*, 2015; Patterson & Bhatt, 2012; Rameshkumar & Rajaram, 2017). Similar impacts have been noted in Venezuela and Panama (Barrios *et al.*, 2007; Sellers *et al.*, 2015). Studies in Hawaii suggest shading by thalli may result in coral death, but these thalli provide substrate for sessile invertebrates (ascidians, sponges) and shelter for holothurians and reef fishes (D. J. Russell, 1983).

Loxothylacus panopaei, a parasitic barnacle native to the Gulf of Mexico and Caribbean Sea, has spread along eastern North America, where it infects the native *Eurypanopeus depressus* (flatback mud crab). Prevalence of infection may reach upwards of 90 per cent in the invaded range (Hines *et al.*, 1997). The parasitic barnacle induces significant behavioural changes, such as reducing mud crab activity, influencing predator-prey relationships, enhancing hiding behaviours and changes in habitat usage in infected crabs. Moreover, infection results in castration of both male and female crabs (Belgrad & Griffen, 2015; Brothers & Blakeslee, 2021; Gehman & Byers, 2017; Toscano *et al.*, 2014).

Mnemiopsis leidyi (sea walnut), native to western Atlantic coastal waters, has spread to European waters (Black Sea, Caspian Sea, Mediterranean, North and Baltic Seas). The earliest records of *Mnemiopsis leidyi* in the Black Sea documented a decrease in mesozooplankton abundance and biomass, changes in diet composition of small pelagic fish, with concomitant reduction in planktivorous fishes (e.g., *Engraulis encrasicolus* (European anchovy)), their eggs and larvae (Finenko *et al.*, 2013, 2015, 2018; Petran & Moldoveanu, 1995; Shiganova, 1997, 1998; Shiganova *et al.*, 2003; Shiganova & Bulgakova, 2000), which were reversed, wholly or partially when *Beroe ovata* (ovate comb jelly), an invasive predator of *Mnemiopsis leidyi*, reduced its population (Finenko *et al.*, 2018; Kamburska *et al.*, 2003; Shiganova *et al.*, 2003). The single study conducted in the Mediterranean Sea documented significant differences in zooplankton abundance in the zooplankton community structure (Fiori *et al.*, 2019). Fearing an outbreak of *Mnemiopsis leidyi* similar to that which had occurred in the Black Sea motivated studies in the North and Baltic Seas (Riisgård *et al.*, 2007). Some studies documented it severely depressed mesozooplankton stocks and influenced bacterioplankton activity and community composition in the vicinity of the jellyfish (Dinasquet *et al.*, 2012; Riisgård *et al.*, 2012). Yet, subsequent studies concluded *Mnemiopsis leidyi* exerted low or no direct predatory pressure on the ecologically important mesozooplankton and ichthyoplankton species and posed no threat to eggs and larvae of commercially important fish such as *Gadus morhua* (Atlantic cod), *Clupea harengus* (Atlantic herring), and *Sprattus sprattus* (European sprat) (Hamer *et al.*, 2011; Jaspers *et al.*, 2011; Javidpour *et al.*, 2009; Kellnreitner *et al.*, 2013; Schaber *et al.*, 2011).

Mytilus galloprovincialis (Mediterranean mussel), native to the Mediterranean and the eastern Atlantic, has been widely introduced both intentionally for cultivation and unintentionally. It is an ecosystem engineer, and dominates wave-exposed rocky shores, increasing invertebrate density and species richness, and changing community composition (Branch *et al.*, 2010; Griffiths *et al.*, 1992; Hanekom & Nel, 2002; T. B. Robinson *et al.*, 2007; T. B. Robinson & Griffiths, 2002). *Mytilus galloprovincialis* has replaced open rocky habitat with complex mussel beds, displacing the native *Choromytilus meridionalis* (black mussel) and native *Scutellastra argenvillei* (Argenville's limpet), but increasing the abundance of *Aulacomya atra* (ribbed mussel) and *Scutellastra granularis* (granular limpet) that now occur within the *Mytilus* beds (Hanekom, 2008; Hanekom & Nel, 2002; Hockey & van Erkom Schurink, 1992; Sadchatheeswaran *et al.*, 2015; Sebastián *et al.*, 2002; Steffani & Branch, 2005). Settling on kelp fronds, *Mytilus galloprovincialis* reduces kelp buoyancy and increases hydrodynamic drag, facilitating uprooting (Lindberg *et al.*, 2020). Its extensive beds provide food for the rare and endangered *Haematopus moquini* (African oystercatcher; Branch & Steffani, 2004; Coleman & Hockey, 2008). In the northeast and northwest Pacific *Mytilus galloprovincialis* has extensively hybridized with *Mytilus trossulus* (northern bay mussel). On the west coast of the United States, hybrids are rare but more frequent near ports and mussel farms (Braby & Somero, 2006; Crego-Prieto *et al.*, 2015; Heath *et al.*, 1995; Rawson *et al.*, 1999; Shields *et al.*, 2010). The hybrid zone in the northwest Pacific runs from the Vladivostok area, Russia, to northern Japan (Brannock & Hilbish, 2010; Skurikhina *et al.*, 2001; Suchanek *et al.*, 1997). Hybridization has also been observed between native southern hemisphere *Mytilus galloprovincialis* and introduced Northeast Atlantic lineages near ports in New Zealand (Gardner *et al.*, 2016).

Pterois volitans (red lionfish), a voracious predator native to the Indo Pacific, has spread to the tropical and subtropical western Atlantic and Caribbean. Its invasion has had significant negative impacts on shallow coral reef fish populations, comprising severe reductions in recruitment, total density, biomass, and species richness of prey-sized fishes, both herbivorous and piscivores (Albins, 2015; Albins & Hixon, 2008; Ingeman, 2016). A shift to an algal dominated community occurred simultaneously with the loss of herbivores, resulting in a decline in corals and sponges at mesophotic depths (Kindinger & Albins, 2017; Lesser & Slattery, 2011). By foraging away from their patch reefs residence, *Pterois volitans* eliminate a spatial refuge from predation used by juveniles of many commercially and ecologically important reef fishes (Benkwitt, 2016; DeRoy *et al.*, 2020).

Pyura praeputialis (cunjuvoi), a solitary tunicate native to Australia, has spread to Chile where it monopolized the low and mid-low rocky intertidal and restricted the native mussel *Perumytilus purpuratus* (purple mussel) to the mid-upper fringe (Caro *et al.*, 2011; Castilla *et al.*, 2004).

Semimytilus patagonicus (bisexual mussel), a mytilid mussel native to the Pacific coast of South America, has spread to southwestern Africa (de Greef *et al.*, 2013; Ma *et al.*, 2020). On rocky, wave-exposed shores *Semimytilus patagonicus* competitively excluded co-occurring mussel species on the low-shore and native species (*Aulacomya atra* (ribbed mussel), *Choromytilus meridionalis* (black mussel)) in the mid-shore, displacing the latter to sublittoral and sand-inundated habitats (Sadchatheeswaran *et al.*, 2015; Skein *et al.*, 2018).

Womersleyella setacea is a turf-forming red alga introduced into the Mediterranean Sea. It has invaded areas where several turf species were absent or evinced lower cover values (Piazzi, Balata, & Cinelli, 2007), causing changes to biodiversity and cover of the epiphytic assemblage of *Posidonia oceanica* (Neptune grass), a species that is endemic to the Mediterranean Sea (Antolić *et al.*, 2008). Some sponge species overgrown by the *Womersleyella setacea* were unable to reproduce, others significantly reduced their reproductive effort (de Caralt & Cebrian, 2013). Following its introduction, colonies of the Mediterranean gorgonian *Paramuricea clavata* (chameleon sea fan), an important structural species in coralligenous assemblages, demonstrated lower survivorship of juvenile colonies, higher necrosis rates and lower biomass (Cebrian *et al.*, 2012).

4.3.3 Documented impacts of invasive alien species on nature by region and taxonomic group

The number of documented negative and positive impacts on nature by invasive alien species varies greatly across regions **(Table 4.12)**.

Negative impacts of invasive alien species across regions

In most regions, plants generally have the greatest number of invasive alien species causing negative impacts **(Table 4.12A)**, except in the Americas, where a large number of invasive alien invertebrates have caused local extinctions (**Table 4.12C**; 41 species), and in Asia-Pacific, where a large number of local extinctions have been caused by invasive alien vertebrates (**Table 4.12B**; 339 documented impacts). *Felis catus* (cat) is responsible for the greatest number of documented local extinctions across all regions (108 records), but mostly in the Asia-Pacific region **(Box 4.11)** and on the Galapagos Islands. Microbes generally have the lowest number of documented impacts across all regions, mostly causing population declines in Europe and Central Asia (**Table 4.12D**; 142 records). The microbe species with the greatest number of documented negative impacts is the oomycete plant pathogen, *Phytophthora ramorum* (38 records), which is known to cause the sudden oak death disease.

Positive impacts of invasive alien species across regions

Positive impacts have been documented in all regions, but the number of invasive alien species with positive impacts (361 species) is substantially lower than the number of species with negative impacts (1623 species). The number of invasive alien plants in the Americas have been documented to be the largest number of invasive alien species with positive impacts, globally (**Table 4.12A**; 109 species). Invasive alien plants in Europe and Central Asia have been documented to cause the greatest number of positive impacts, globally (**Table 4.12A**; 406 records).

The invasive alien plant with the greatest documented number of positive impacts on nature is *Robinia pseudoacacia* (black locust; 55 records), often resulting in increase in abundance and richness of native pollinators attracted to the abundant production of nectar by this alien plant. *Robinia pseudoacacia* (black locust) also has 44 documented negative impacts on nature (Vítková *et al.*, 2017).

Dreissena polymorpha (zebra mussel) is the invasive alien species with the greatest number of positive documented impacts on native species (143 impacts). *Dreissena polymorpha* has positive impacts on a wide range of native species, mostly invertebrates, through water filtering, thereby changing water chemistry and turbidity, which in turn favours littoral invertebrate communities, but disfavours planktonic communities (Strayer, 2009). *Dreissena polymorpha* is also in the top ten invasive alien invertebrate species causing negative impacts (85 records), and the nature of the invasion by the species (particularly in North America) and the conflicting interpretation of its impacts has been well documented (Strayer, 2009).

Native species impacted by invasive alien species across regions

Native plant species are generally the most often negatively affected taxa across all regions. However the large number of local extinctions of native vertebrates in Asia-Pacific (**Table 4.13B**; 284 records) and of native invertebrate species in the Americas and Asia-Pacific regions **(Table 4.13C)** constitute exceptions to this general pattern. *Linepithema humile* (Argentine ant) has been documented to cause the greatest number of local extinctions of

Table 4 12 **Number of invasive alien species causing positive and negative impacts on nature by region.**

The number of plants A), vertebrates B), invertebrates C), microbes D) causing negative and positive impacts by region and by taxa. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

A) Plants: Number of invasive alien species (number of impacts)

Region	Negative impacts caused by invasive alien plants				Positive impacts
	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	
Africa	131 (576)	83 (153)	65 (184)	8 (31)	10 (28)
Americas	408 (2494)	151 (393)	196 (727)	21 (48)	109 (337)
Asia-Pacific	246 (1034)	182 (364)	109 (307)	19 (52)	42 (103)
Europe and Central Asia	129 (3767)	47 (174)	103 (805)	12 (55)	46 (406)
Antarctica		1 (1)			

B) Vertebrates: Number of invasive alien species (number of impacts)

Region	Negative impacts caused by vertebrates				Positive impacts
	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	
Africa	37 (132)	37 (93)	31 (107)	13 (45)	2 (3)
Americas	49 (576)	101 (313)	60 (360)	30 (196)	17 (45)
Asia-Pacific	139 (1589)	117 (505)	92 (620)	37 (339)	21 (58)
Europe and Central Asia	31 (138)	76 (222)	39 (92)	22 (39)	5 (11)
Antarctica	1 (4)		1 (2)		1 (1)

C) Invertebrates: Number of invasive alien species (number of impacts)

Region	Negative impacts caused by vertebrates				Positive impacts
	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	
Africa	67 (397)	30 (58)	8 (45)	6 (39)	4 (37)
Americas	241 (1046)	81 (451)	86 (407)	41 (154)	37 (400)
Asia-Pacific	92 (522)	75 (212)	67 (176)	26 (117)	25 (57)
Europe and Central Asia	237 (1196)	43 (150)	45 (226)	25 (83)	34 (169)

D) Microbes: Number of invasive alien species (number of impacts)

Region	Negative impacts by microbes				Positive impacts
	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	
Africa	23 (45)	1 (1)			
Americas	26 (125)	4 (9)	17 (58)	4 (10)	2 (3)
Asia-Pacific	11 (18)	11 (15)	9 (17)	3 (4)	
Europe and Central Asia	16 (189)	7 (44)	12 (142)	1 (1)	1 (1)

native invertebrate species across all regions, mostly by outcompeting native ants, but also through predation on native invertebrates. Native microbes generally have the lowest number of documented impacts by invasive alien species across all regions, with the highest number of negative impacts being native microbe population declines in Europe and Central Asia (**Table 4.13D**; 24 records). The perennial woody shrub *Rosa rugosa* (rugosa rose) has caused the greatest number of documented population declines in native microbes (5 records), through changes in soil chemistry, particularly in coastal dune habitats (Stefanowicz *et al.*, 2019).

Table 4 13 **Number of invasive alien species causing impacts on native taxa by region.**

The number of invasive alien species impacting A) native plants, B) vertebrates, C) invertebrates and D) microbes and the number of documented impacts (in brackets) in each region. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

A) Number of invasive alien species impacting native plants (number of impacts)

Region	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	Positive impacts
Africa	203 (610)	103 (159)	68 (99)	8 (8)	1 (2)
Americas	496 (1554)	187 (397)	240 (675)	28 (81)	76 (130)
Asia-Pacific	287 (884)	232 (478)	134 (320)	27 (64)	16 (20)
Europe and Central Asia	269 (2030)	67 (216)	114 (620)	22 (58)	11 (30)

B) Number of invasive alien species impacting native vertebrates (number of impacts)

Region	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	Positive impacts
Africa	87 (209)	37 (67)	68 (112)	10 (38)	6 (19)
Americas	124 (679)	98 (310)	90 (383)	32 (142)	36 (93)
Asia-Pacific	169 (1189)	111 (356)	105 (469)	35 (284)	40 (89)
Europe and Central Asia	53 (293)	70 (188)	51 (114)	17 (28)	19 (71)
Antarctica	1(4)		1 (2)		1 (1)

C) Number of invasive alien species impacting native invertebrates (number of impacts)

Region	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	Positive impacts
Africa	55 (171)	24 (29)	38 (87)	10 (37)	12 (41)
Americas	225 (996)	109 (329)	110 (379)	44 (162)	58 (439)
Asia-Pacific	123 (532)	73 (161)	89 (257)	36 (154)	36 (81)
Europe and Central Asia	159 (1501)	64 (139)	90 (477)	24 (86)	34 (180)
Antarctica		1 (1)			

D) Number of invasive alien species impacting native microbes (number of impacts)

Region	Ecosystem impacts	Impacts on individuals	Population declines	Local extinction	Positive impacts
Africa	4 (4)	1 (1)			
Americas	37 (83)	12 (21)	12 (16)	2 (4)	8 (14)
Asia-Pacific	15 (27)	3 (4)	11 (13)	1 (1)	6 (9)
Europe and Central Asia	25 (140)		9 (24)		7 (10)

Box 4 12 Impacts of fox and cat predation in Australia.

Multiple studies have established that *Felis catus* (cat) and *Vulpes vulpes* (red fox) have had particularly significant impacts on, and continue to threaten, many native Australian vertebrate species (Doherty *et al.*, 2016; Hunter *et al.*, 2018; Radford *et al.*, 2018; Saunders *et al.*, 2010; Woinarski *et al.*, 2015). *Vulpes vulpes* has been shown to suppress populations of *Petrogale lateralis* (black-footed rock-wallaby; Kinnear *et al.*, 1988, 1998; **Figure 4.24**), *Dasyurus geoffroii* (western quoll; Morris *et al.*, 2003), ground-dwelling and arboreal mammals (Hunter *et al.*, 2018), medium-sized marsupials (Dexter & Murray, 2009), and even large species such as *Macropus giganteus* (eastern grey kangaroo; Banks *et al.*, 2000). When not controlled, *Vulpes vulpes* also reduce abundance of *Varanus gouldii* (sand goanna), diurnal scincid lizards (Olsson *et al.*, 2005), and *Varanus varius* (lace monitor; Hu *et al.*, 2019), and can destroy turtle nests, severely impacting their populations (R.-J. Spencer *et al.*, 2006; Limpus & Reimer, 1994). *Vulpes vulpes* have also been implicated in colonial seabird (Norman, 1971), *Leipoa ocellata* (malleefowl; Wheeler & Priddel, 2009; S. L. Williams, 1995), and ground-foraging passerine declines (Ford *et al.*, 2001).

Felis catus is currently considered the single most significant threat to Australian mammals (Frank *et al.*, 2014; Woinarski *et al.*, 2015). Indeed, *Felis catus* has been implicated in approximately two thirds of Australian native mammal extinctions, and another 54 native mammal taxa have suffered severe range contractions and are seriously threatened by cat predation. *Felis catus* caused local extirpation of a native rodent, *Rattus villosissimus* (long-haired rat) in a Northern Territory tropical savanna (Frank *et al.*, 2014), and has been identified as a factor contributing to northern Australian mammal declines (Woinarski *et al.*, 2011; D. O. Fisher *et al.*, 2014). Davies *et al.* (2017) demonstrated that *Felis catus* predation on threatened *Conilurus penicillatus* (brush-tailed rabbit-rat) is driving the remnant population to extinction on Melville Island, suggesting that predation has likely been a significant driver of *Conilurus penicillatus* decline throughout northern Australia **(Figure 4.24)**. Additionally, predation of juvenile *Dasyurus viverrinus* (eastern quoll) by *Felis catus* is likely inhibiting recovery of low-density quoll populations across Tasmania (Fancourt *et al.*, 2015).

Figure 4 24 **Examples of native species with serious population declines due to invasive alien species.**

Petrogale lateralis (black-footed rock-wallaby, left), *Leipoa ocellata* (malleefowl, middle), *Conilurus penicillatus* (brush-tailed rabbit rat, right). Photo credit: Kym Nicolson, WM Commons – CC BY 4.0 (left) / butupa, WM Commons – CC BY 2.0 (middle) / Hugh Davies – CC BY 4.0 (right).

There is clear evidence to implicate predation by *Felis catus* in the loss of wildlife populations at a local and regional scale, but the contribution of *Felis catus* predation to Australian extinctions or extirpations is hard to disentangle from confounding other threats such as habitat clearance, changing fire regimes, and other feral vertebrates. On offshore islands, where confounding factors are less severe, *Felis catus* have been shown to decimate native fauna (D. C. Duffy & Capece, 2012). Predation has caused the extinction of *Cyanoramphus novaezelandiae erythroti* (Macquarie Island Parakeet; R. H. Taylor, 1979), *Traversia lyalli* (Stephens Island wren; Galbreath, 2004), and the extirpation from Marion Island of *Pelecanoides urinatrix* (common diving petrel; Bloomer & Bester, 1991; Cooper *et al.*, 1995).

In addition to direct extinction of species, *Felis catus* predation can have significant knock-on effects at the ecosystem level, through alteration of ecosystem functioning. The local extinction of fossorial mammals (i.e., those that dig burrows underground), in Australian arid and semi-arid regions (Tuft *et al.*, 2021; Doherty *et al.*, 2017) has caused a loss of key soil-engineering processes, negatively impacting associated plant communities (James & Eldridge, 2007; Eldridge & James, 2009; James *et al.*, 2011).

4.4 IMPACTS OF BIOLOGICAL INVASIONS ON NATURE'S CONTRIBUTIONS TO PEOPLE

4.4.1 General patterns

Globally, the impact database collected through this chapter contains 6,211 impacts of invasive alien species on nature's contributions to people. The economic costs of invasive alien species are presented in **Box 4.13**. Impacts on nature's contributions to people can be negative or positive (**section 4.1.2**); they are considered negative when humans are harmed and positive when humans benefit from changes in nature's contributions to people by invasive alien species. In total, there is evidence of 4,905 negative impacts (78.9 per cent of all impacts on nature's contributions to people) caused by 1,337 invasive alien species on all nature's contributions to people categories, indicating the multiplicity of impacts that invasive alien species can have beyond nature (Vilà *et al.*, 2010). There are also 421 invasive alien species that have caused 1306 positive impacts (20.8% of all impacts on nature's contributions to people; **Figure 4.28**).

Box 4 13 The economic costs of biological invasions.

There are many case studies of economic costs of biological invasions worldwide (Dana *et al.*, 2014; Diagne, Leroy, *et al.*, 2020). As a result, these costs display a very high degree of heterogeneity (e.g., nature, origin, type, implementation, estimation approach, spatial and temporal scales) and lack standardized methods that would have allowed relevant compilations and comparisons, which in turn may provide key insights for management actions (Diagne, Catford, *et al.*, 2020). In addition, as costs are most often provided at the local scale, global estimations are very scarce while biological invasions still remain an increasingly planet-wide issue (Diagne, Catford, *et al.*, 2020; Diagne, Leroy, *et al.*, 2021; Latombe *et al.*, 2017; Pagad *et al.*, 2018). Consequently, the only global figures available up to recently were based on a handful of studies that used crude extrapolations from individual estimations (Kettunen *et al.*, 2009; Pimentel *et al.*, 2001), much criticized by ecologists and economists alike (e.g., Bradshaw *et al.*, 2016; Hoffmann & Broadhurst, 2016; T. P. Holmes *et al.*, 2009). Yet, these pioneer studies had the merit to suggest very high economic costs, and then trigger more robust assessments on many taxa or regions, as well as some more robust, global estimates, for example on a given economy sector (Paini *et al.*, 2016). Recently, the InvaCost project[4] has compiled a wealth of individual cost estimates in a public and updatable database and has devised a standardized method of calculating economic costs of biological invasions (Diagne, Leroy, *et al.*, 2020). This allows many comprehensive analyses of this particular dimension of impacts of biological invasions (Diagne, Catford, *et al.*, 2020).

The main results of these analyses (see Diagne, Leroy, *et al.*, 2021 for the very first analysis) are that (i) the global economic costs of biological invasions over the last 50 years (1970-2019) are massive, at least US$1,738 billion[5] if only the most robust data are taken into account **(Figure 4.26)**; (ii) these costs are increasing exponentially with a four-fold increase each decade[6] **(Figure 4.25)**; (iii) being based on already published and collated studies, the costs are massively underestimated (for example, currently occurring costs are not yet documented for most economically harmful invasive alien species and invaded countries) and (iv) management expenditures represent a very small fraction of the total costs, with damage cost recently shown to constitute 92 per cent of the total cost estimated (Cuthbert *et al.*, 2022). In 2017 alone, aggregate global invasive alien species invasion costs were estimated to reach until US$162.7 billion, exceeding the 2017 gross domestic product (GDP) of 52 of the 54 countries on the African continent, and more than twenty times higher than the combined total funds available in 2017 for the World Health Organization (WHO) and the United Nations (Diagne, Leroy, *et al.*, 2021). Applying a similar method (Leroy *et al.*, 2022) to the most up-to-date version of InvaCost (at the time of writing this report) has led to an upper prediction of US$423.3 billion for the year 2019.[7]

There are now a number of published analyses from this database. For now, they are mostly descriptive and synthesize economic costs of invasions in different regions of the world or from different invasive alien taxonomic groups. Studies with a geographical focus have shown that reported costs are more important in some regions, such as North America (Crystal-Ornelas *et al.*, 2021) and Asia (Liu *et al.*, 2021), and less so in regions such as Europe (Haubrock, Turbelin, *et al.*, 2021), Africa (Diagne, Turbelin, *et al.*, 2021) and South and Central America (Heringer *et al.*, 2021), most probably due to knowledge gaps. Studies have also been conducted at the country level: in Argentina (Duboscq-Carra *et al.*, 2021), Australia (Bradshaw *et al.*, 2021), Brazil (Adelino *et al.*, 2021), Canada (Vyn, 2022), Ecuador (Ballesteros-Mejia *et al.*, 2021), France (Renault *et al.*, 2021), Germany (Haubrock, Cuthbert, Sundermann, *et al.*, 2021), India (Bang *et al.*, 2022), Japan (Watari *et al.*, 2021),

4. www.invacost.fr
5. Equivalent 2017 US$.
6. Based on new analyses using the latest version of the InvaCost database (version 4.0) available at the time of writing this report (Leroy *et al.*, 2022, 2021).
7. Data management report available at https://doi.org/10.5281/zenodo.7857828

Box 4 13

Mexico (Rico-Sánchez *et al.*, 2021), New Zealand (Bodey *et al.*, 2022), Russia (Kirichenko *et al.*, 2021), Singapore (Haubrock, Cuthbert, Yeo, *et al.*, 2021), Spain (Angulo, Ballesteros-Mejia, *et al.*, 2021) the United Kingdom (Cuthbert, Bartlett, *et al.*, 2021) and in the United States (Fantle-Lepczyk *et al.*, 2021). Interestingly, the large number of countries already surveyed show both commonalities (such as high, underestimated and increasing costs) and specificities (such as the costliest species, the most impacted sectors or the proportion of management expenditures *versus* damage costs).

Studies focusing on the economic impact of particular taxonomic groups are fewer for now, mostly due to a lack of reported costs in the literature, but the following have sufficiently high cost data to have warranted dedicated studies: fishes (Haubrock *et al.*, 2022), bivalves (Haubrock, Cuthberg, *et al.*, 2021), crayfishes and crabs (Kouba *et al.*, 2021), terrestrial invertebrates (Renault *et al.*, 2022), aquatic species (Cuthbert, Pattison, *et al.*, 2021), or ants (Angulo *et al.*, 2022). As cost data accumulate, other syntheses are being prepared.

Because InvaCost is a living database, it is regularly being updated, and published studies based on it may refer to earlier versions, with actual costs having increased since then. For this reason, a "living figure", directly linked to the latest version of the database and automatically updated, is available online (Leroy *et al.*, 2021). In addition, different studies have used different strategies regarding the filtering steps of their dataset processing. As a consequence, cost estimates highlighted in those studies may not necessarily be comparable. For example, most (but not all) studies focused on "observed" costs and those classified as "highly reliable" from a methodological point of view **(Figure 4.26)**. As a result, all the cost estimates provided should be considered as relative orders of magnitude, which remains a good indication of both the reported costs and the knowledge gaps.

From the living cost figure, in which all costs filters are identical, and which therefore allows meaningful comparisons, one can assess the costliest invasive alien species and the most impacted invaded regions, as they are currently reported in the literature **(Figure 4.27)**.

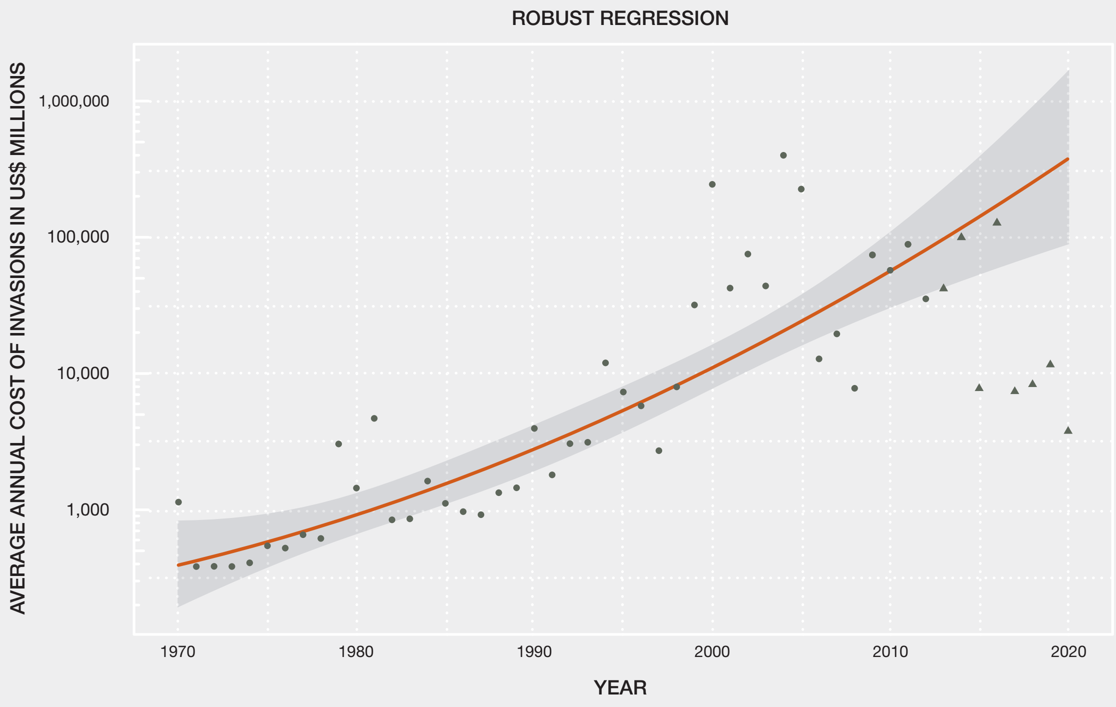


Figure 4 25 **Temporal trend of global invasion costs (in millions of 2017 US$) between 1970 and 2019.**

A model prediction approach based on an Ordinary Least Square (OLS) regression was used in order to take into account (i) the dynamic nature of costs, (ii) the time lags between the real occurrence of the costs and their reporting in the literature (called "publication delay" hereafter), (iii) the heteroscedastic and temporally auto-correlated nature of cost data, and (iv) the effects of potential outliers in the cost estimates. The model was calibrated and fitted with at least 75 per cent of cost data completeness. All methodological details necessary for the rationale behind model selection as well as for obtaining this figure are presented in Diagne, Leroy *et al.* (2021), Leroy *et al.* (2022). Data management report available at https://doi.org/10.5281/zenodo.7857828

Box 4 13

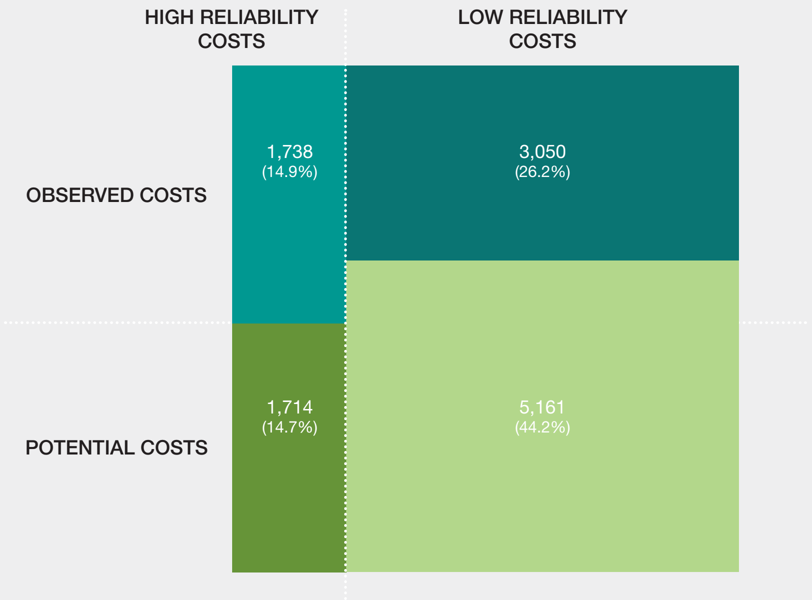
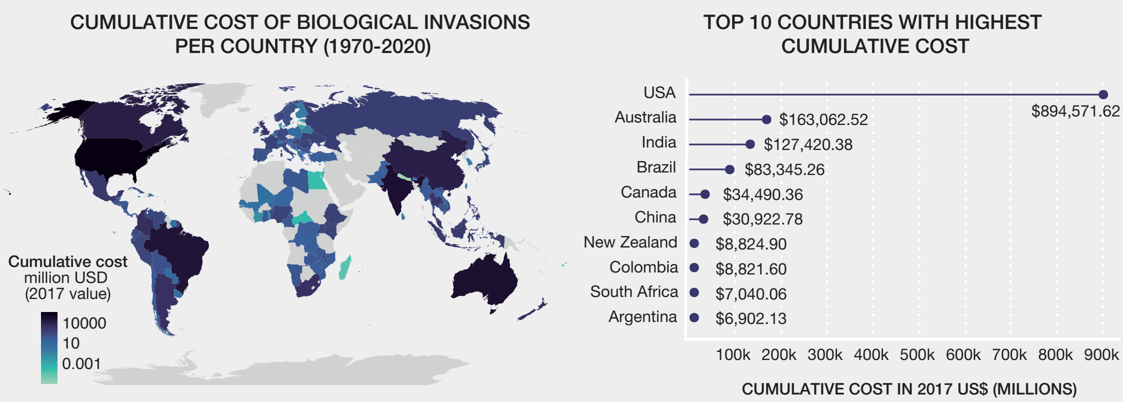


Figure 4 26 **The proportion of costs in the InvaCost database according to their implementation and reliability.**

Numbers in the square represent the total cost in US$ billion, and the corresponding percentage of the whole in parentheses. All costs have been standardized in equivalent 2017 US$ (Diagne, Leroy, *et al.*, 2020 for methodological details). InvaCost displays over 13,000 individual costs (at the time of writing this report), each described with 64 variables characterizing the record, the study, the typology of the economic cost and the invasive alien species. Among these, two are of major importance: implementation, i.e., whether the costs are actually observed or extrapolated and predicted (called "potential") and whether the original methodology led to a classification into either high or low reliability. The choice of these two variables dictates the number of costs accounted for in different studies, the resulting final global estimate and its overall robustness. This chapter only considers the most robust subset of InvaCost, the costs that are simultaneously observed and of high reliability (upper left square, less than 15 per cent of all available data). Note that this figure represents data recorded in the latest version of the InvaCost database available at the time of writing this report, and the proportions displayed here are likely to evolve as the database is updated over time. All cost information are regularly updated (Leroy *et al.*, 2021).



Box 4 13

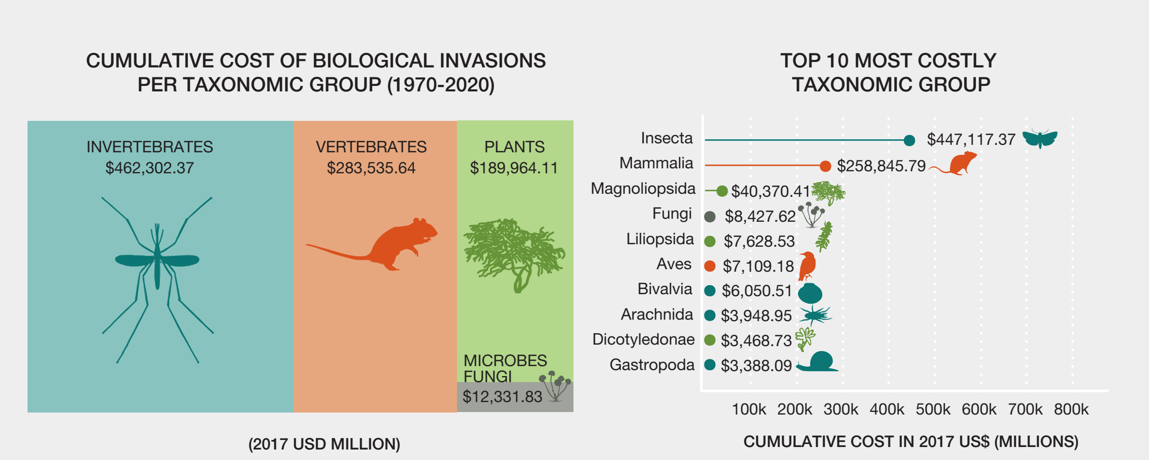


Figure 4 27 **Synthesis of cumulative economic costs of biological invasions.**[8]

As available in the literature and standardized in the InvaCost database (latest version 4.0 available at the time of writing this report): for all countries in the world (left on the previous page), the 10 countries with the highest cumulative costs (right on the previous page) and the four major taxonomic groups (left) as well as the ten costliest taxa (right). All costs have been standardized in equivalent 2017 US$ (Diagne, Leroy, *et al.* (2021) and Leroy *et al.* (2022) for methodological details) and only the most robust subset has been used here **(Figure 4.26)**. Note that this figure represents data recorded in the latest version of the InvaCost database available at the time of writing this report, and the proportions displayed here are likely to evolve as the database is updated over time. All cost information are regularly updated (Leroy *et al.*, 2021 for the most up-to-date figures).
Data management report available at https://doi.org/10.5281/zenodo.8231570

Most impacted categories of nature's contributions to people

More than 66 per cent of documented impacts on nature's contributions to people are on the provision of food and feed **(Figure 4.28)**. These include mainly decreases in crop and forest tree production caused by alien weeds, pests and pathogens (Fried *et al.*, 2017; Kenis *et al.*, 2017), but also the impact of invasive alien microbes on livestock (French, 2017) and the impact of invasive alien species on fisheries and aquaculture (Gozlan, 2017). Most invasive alien species cause negative impacts on provision of food and feed (748 species), on habitat creation and maintenance (255 species) and on provision of materials, companionship and labour (301 species). Invasive alien species also cause positive impacts on provision of food and feed (199 species), on medicinal, biochemical and genetic resources (83 species) and on the formation, protection and decontamination of soils and sediments (77 species).

8. The boundaries and names shown, and the designations used on the maps shown here do not imply official endorsement or acceptance by IPBES.

Conflict species causing both positive and negative impacts

There are some invasive alien species that cause both positive and negative impacts on nature's contributions to people, which causes conflicts among different socioeconomic sectors as, for instance, the farming and conservation sectors (Vilà & Hulme, 2017). This duality makes the quantification of nature's contributions to people a challenge. Conflicting values are prominently found with respect to invasive alien trees which are seen as positive because they provide wood and contribute to carbon sequestration and thus to climate regulation; however, at the same time, many alien trees increase fire hazards (Castro-Díez *et al.*, 2019) and decrease the recreational use of forests (Vaz *et al.*, 2018; **Chapter 5**, **section 5.6.1.2**).

Invasive alien species most often documented causing impacts on nature's contribution to people

The top ten species that are most often documented to have negative impacts on nature's contributions to people comprise four plants, five invertebrates and a fish **(Table 4.14A)**. *Pontederia crassipes* (water hyacinth) and many other aquatic plants have pervasive impacts on water

Figure 4 28 **Documented numbers of invasive alien species causing negative and positive impacts on categories of nature's contributions to people.**

Positive and negative stacked bar charts do not imply that positive and negative impacts can be summed. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

quality and quantity, clog irrigation and draining ditches, and thereby interfere with boating and fishing (Brundu, 2015; Ueki *et al.*, 1976). The plants *Reynoutria japonica* (Japanese knotweed) and *Impatiens glandulifera* (Himalayan balsam), and the tree *Robinia pseudoacacia* (black locust), which commonly invade central European habitats, cause impacts on soil quality and on pollination (Dassonville *et al.*, 2011; Nienhuis *et al.*, 2009). *Dreissena polymorpha* (zebra mussel), one of the most studied freshwater invertebrates, has negative impacts on nature's contributions to people by, for example, disrupting energy production (Ludyanskiy *et al.*, 1993; Karatayev *et al.*, 2005). Four invertebrates affect food and feed provision: *Solenopsis invicta* (red imported fire ant), *Bactrocera dorsalis* (Oriental fruit fly), *Chilo partellus* (spotted stem borer), and *Lissachatina fulica* (giant African land snail). *Cyprinus carpio* (common carp) is the invasive alien vertebrate with most documented impacts; it eats submerged vegetation and destroys hatching grounds for small native fishes, invertebrates, or other aquatic animals, changes the nutrient compositions in water through grubbing sediments (Matsuzaki *et al.*, 2009), and spreads Koi herpesvirus to native carp populations that show higher mortality than introduced populations (Uchii *et al.*, 2009).

The top ten invasive alien species that are most often documented to have positive impacts on nature's contributions to people are plants **(Table 4.14B)**, all of which also cause negative impacts. For example, *Acacia longifolia* (golden wattle) and *Acacia dealbata* (acacia bernier) are N-fixing species that have been introduced to restore degraded soils but at the same time, their invasion modifies the structure of the habitats to be both beneficial or detrimental to people depending on their socioeconomic and cultural context (Kull *et al.*, 2011). Similarly, *Prosopis juliflora* (mesquite) affects the availability of fodder for domestic livestock by reducing grassland area and grass cover (P. N. Joshi *et al.*, 2009; Kohli *et al.*, 2006; Timsina *et al.*, 2011), but at the same time constitutes an important source of fuelwood (Dayal, 2007; Duenn *et al.*, 2017), its stems can be used for fencing (D. Bartlett *et al.*, 2018; Duenn *et al.*, 2017), it can improve soil quality *via* biochar (D. Bartlett *et al.*, 2018), and there are reports of people adapting to the use of plant parts for medicinal purposes (Duenn *et al.*, 2017). The overwhelming negative impacts of *Prosopis juliflora* on nature's contributions to people are not offset by its positive impacts.

Table 4 14 **Top 10 invasive alien species with most documented negative and positive impacts on nature's contributions to people.**

The invasive alien species with most documented A) negative and B) positive impacts on nature's contributions to people. Note that this is not an indication of the global impact of these species, but of the number of cases found and analysed in this report. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate: Microorganisms:

A) Negative impacts on nature's contributions to people

Species	Taxa	Nature's contributions people (number of documented impacts)
Pontederia crassipes (water hyacinth)		Energy (2); Food & feed (32); Freshwater quantity (19); Options (2); Physical experiences (4); Water quality (18)
Solenopsis invicta (red imported fire ant)		Biological processes (13); Energy (3); Food & feed (35); Learning (1); Materials (12); Options (4)
Dreissena polymorpha (zebra mussel)		Energy (17); Freshwater quantity (4); Materials (13); Medicinal (2); Ocean acidification (1); Options (8); Water quality (7)
Bactrocera dorsalis (Oriental fruit fly)		Food & feed (41)
Impatiens glandulifera (Himalayan balsam)		Biological processes (9); Freshwater quantity (4); Pollination & dispersal (5); Soils formation (22)
Robinia pseudoacacia (black locust)		Biological processes (13); Soils formation (27)
Chilo partellus (spotted stem borer)		Food & feed (37)
Lissachatina fulica (giant African land snail)		Food & feed (36)
Reynoutria japonica (Japanese knotweed)		Soils formation (33)
Cyprinus carpio (common carp)		Food & feed (28)

B) Positive impacts on nature's contributions to people

Species	Taxa	Nature's contributions people (number of documented impacts)
Solidago gigantea (giant goldenrod)		Climate (6); Soils formation (48)
Robinia pseudoacacia (black locust)		Energy (6); Food & feed (10); Materials (2); Physical experiences (8); Soils formation (26)
Acacia longifolia (golden wattle)		Climate (5); Freshwater quantity (3); Hazards (1); Soils formation (37)
Impatiens glandulifera (Himalayan balsam)		Biological processes (4); Freshwater quantity (5); Pollination & dispersal (8); Soils formation (23)
Reynoutria japonica (Japanese knotweed)		Climate (3); Pollination (3); Soils formation (32)
Rosa rugosa (rugosa rose)		Biological processes (6); Climate (2); Hazards (1); Physical experiences (3); Soils formation (21)
Prosopis juliflora (mesquite)		Energy (9); Food & feed (10); Habitat (2); Materials (6); Medicinal (3); Physical experiences (2)
Acacia dealbata (acacia bernier)		Climate (4); Soils formation (23)
Carpobrotus edulis (hottentot fig)		Air quality (1); Freshwater quantity (3); Hazards (2); Soils formation (15)
Pontederia crassipes (water hyacinth)		Biological processes (4); Food & feed (5); Hazards (1); Materials (2); Medicinal (3); Water quality (2)

4.4.1.1 Islands vs. mainland

Despite the seminal and substantial body of literature on the threats and impacts of invasive alien species in remote islands, such as Hawaii, the Galapagos or New Zealand, only 12 per cent of the impacts on nature's contributions to people are documented on islands. The vast majority (76.4 per cent) of impacts on islands are negative **(Figure 4.29)** which is similar to the proportion of negative impacts on mainlands (79.3 per cent). Also, the affected categories of nature's contributions to people for which there are the most documented impacts are similar between islands and mainlands: namely on provision of food and feed (caused by 139 invasive alien species), provision of materials, companionship and labour (51 invasive alien species), and on habitat creation and maintenance (44 invasive alien species). The proportion of documented positive impacts is more important on islands than on mainlands, noticeably on food and feed (44.8 per cent on islands against 22.2 per cent on mainlands) and pollination and propagule dispersal (10.4 per cent on islands against 3 per cent on mainlands). On the contrary, on islands, the proportion of impacts on the formation, protection and decontamination of soils and sediments (17.5 per cent on islands against 35.8 per cent on mainlands), medicinal, biochemical and genetic resources (2.2 per cent on islands against 9.8 per cent on mainlands) are smaller than on mainlands. Biogeographic comparative analysis between homologous habitats in islands and mainland invaded by the same invasive alien species are necessary to identify the consistency in direction and intensity of their impacts on nature's contributions to people (D'Antonio & Dudley, 1995).

4.4.1.2 Protected areas

The SCOPE international programme on biological invasions indicated the need for research, monitoring (**Glossary**) and management of the impacts of invasive alien species in

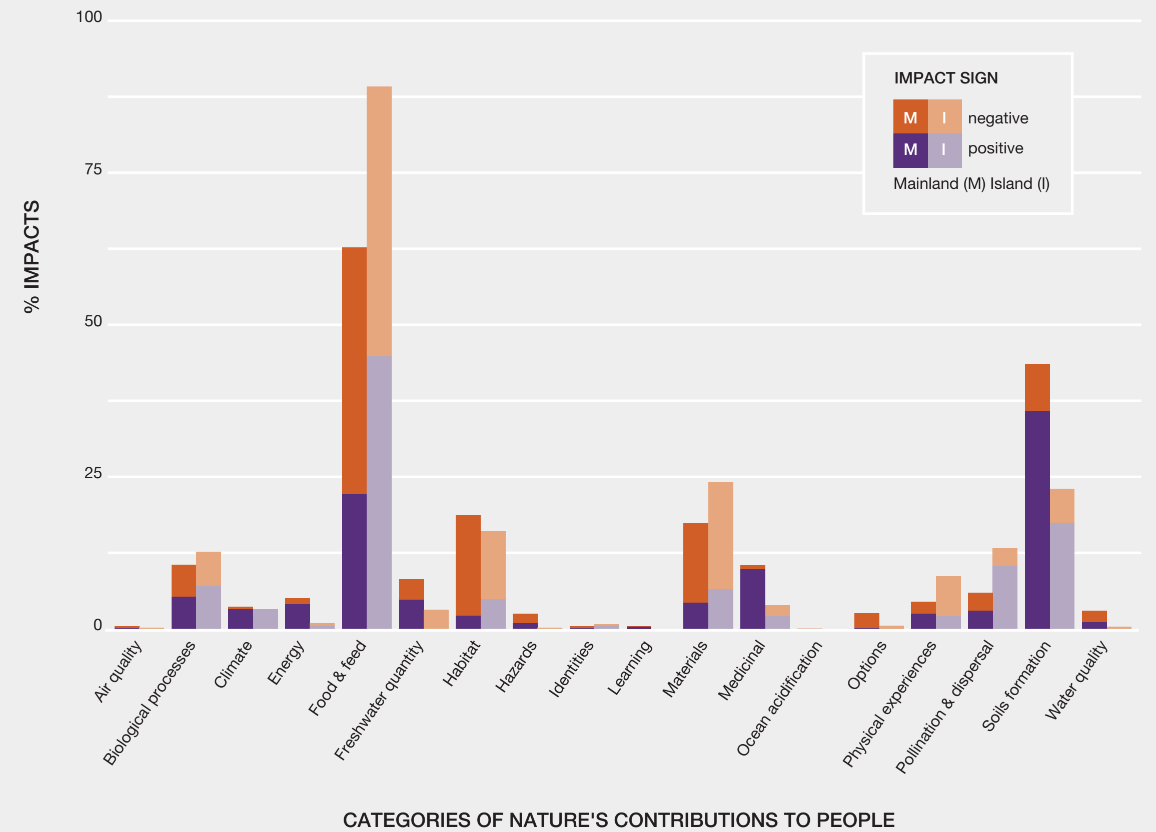


Figure 4 29 **Negative and positive impacts on nature's contributions to people on mainland and on islands.**

This figure shows the percentage (y axis) of positive (bottom half of bar) and negative impacts (top half of bar) on islands and on mainland or unknown territories for each category of nature's contributions to people (x axis). Positive and negative stacked bar charts do not imply that positive and negative impacts can be summed. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069 protected areas (Foxcroft, Pyšek, *et al.*, 2013). However, a recent analysis has shown that the perceived threat in protected areas has worsened over time especially for plants (Foxcroft, Pyšek, *et al.*, 2013; R. T. Shackleton *et al.*, 2020). Five per cent of the documented impacts on nature's contributions to people occur in protected areas, with more positive (54.7 per cent) than negative (45.3 per cent) impacts. More than 50 per cent of the documented impacts in protected areas concern changes of the formation, protection and decontamination of soils and sediments. Other important impacts are on the provision of food and feed (13.2 per cent of the impacts on nature's contributions to people in protected areas), the regulation of freshwater quantity, location and timing (8.4 per cent) and regulation of detrimental organisms and biological processes (8.4 per cent). Notwithstanding, the importance of protected areas for their cultural, sometimes sacred value, there are no documented impacts on non-material nature's contributions to people such as impacts on learning and inspiration, physical and psychological experiences and supporting identities.

4.4.2 Documented impacts of invasive alien species on nature's contributions to people by realm

4.4.2.1 Patterns of negative and positive impacts of invasive alien species on nature's contributions to people in the terrestrial realm

Impacted units of analysis in the terrestrial realm

There are many more documented negative impacts of invasive alien species on nature's contributions to people than positive impacts (3,424 against 1,103, respectively) in the terrestrial realm. Cultivated areas (approximately 33 per cent of impacts) and temperate and boreal forests (approximately 20 per cent of impacts) together account for more than half of all documented negative impacts of invasive alien species on nature's contributions to people. These are followed by urban/semi-urban areas, temperate grasslands, and tropical and subtropical dry and humid forests (8 to 10 per cent each). The remaining four units of analysis together constitute less than 20 per cent of all documented impacts. In the case of positive impacts, the largest proportion of documented impacts is from temperate and boreal forests (23 per cent), followed by temperate grassland (approximately 19 per cent). Cultivated areas, tropical and subtropical dry and humid forests, and Mediterranean woodlands, forests and scrub account for between 11 and 15 per cent each. The rest of the four terrestrial units of analysis together constitute less than one-fifth of all documented impacts. The predominance of certain units of analysis amongst documented impacts of invasive alien species on nature's contributions to people is more likely a reflection of information availability rather than actual impacts. For example, in a recent review of alien trees and their impacts on ecosystem services, Castro-Díez *et al.* (2019) found that the temperate and Mediterranean biomes were over-represented proportionate to their area, compared with other large regions of the world, such as Asia and Africa.

Impacted categories of nature's contributions to people in the terrestrial realm

Negative impacts of invasive alien species on nature's contributions to people are dominated by the categories of food and feed (40 per cent of documented impacts) followed by habitat creation and maintenance (approximately 20 per cent of documented impacts). Negative impacts on the category, formation, protection and decontamination of soils and sediments account for an additional 9 per cent of documented impacts, with all other categories together constituting less than 30 per cent of documented negative impacts **(Figure 4.30)**. Positive impacts are dominated by impacts on the categories, formation, protection and decontamination of soils and sediments (38 per cent), food and feed (21 per cent), and medicinal, biochemical and genetic resources (approximately 10 per cent). The remaining categories of nature's contributions to people together account for only 30 per cent of all documented positive impacts **(Figure 4.31)**.

Negative impacts on particular categories of nature's contributions to people predominate in certain units of analysis **(Figure 4.30)**. For example, negative impacts on food and feed are prominent in cultivated areas (76 per cent of documented impacts, caused by 321 species). Negative impacts on food and feed are also a large proportion of total documented negative impacts in tropical and subtropical dry and humid forests (45 per cent of documented impacts caused by 55 species), tropical and subtropical savannas and grasslands (32 per cent of documented impacts caused by 49 species) and urban/semi-urban areas (about 30 per cent of documented impacts caused by 42 species).

Impacts on habitat creation and maintenance account for about 40 to 50 per cent of all documented negative impacts in deserts and xeric shrublands (50 per cent of documented impacts, caused by 29 species), temperate grasslands (43 per cent of documented impacts caused by 74 species), and tropical and subtropical savannas and grasslands (36 per cent of documented impacts caused by 31 species). So also, impacts on formation, protection and decontamination of soils and sediments constitute between a fifth to a fourth of all negative impacts in Mediterranean woodlands forests and scrub (27 per cent of documented

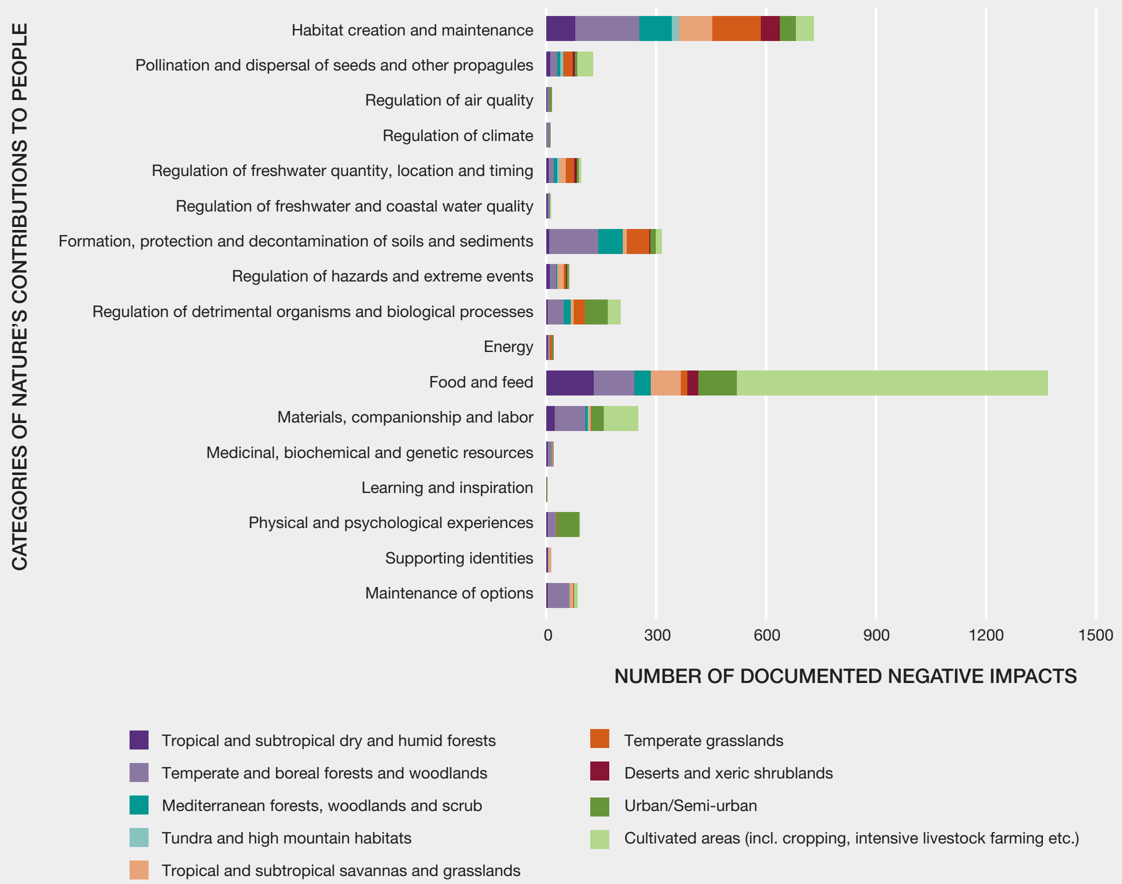


Figure 4 30 **Documented number of negative impacts of invasive alien species (x axis) on categories of nature's contributions to people (y axis) across different terrestrial units of analysis.**

There are 3,424 documented negative impacts across all categories of nature's contributions to people in the terrestrial realm. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

impacts caused by 22 species), followed by temperate grasslands (20 per cent of documented impacts caused by 19 species), and temperate and boreal forests (19 per cent of documented impacts caused by 25 species).

As with negative impacts of invasive alien species, certain types of positive impacts on nature's contributions to people also predominate in particular units of analysis **(Figure 4.31)**. Impacts on the category, formation, protection and decontamination of soils and sediments, outweigh positive impacts on all other categories of nature's contributions to people in Mediterranean woodlands forests and scrub (70 per cent of documented impacts; 14 species). Positive impacts on formation, protection and decontamination of soils and sediments also account for between a third to half of all documented impacts in temperate grassland (53 per cent of documented impacts, caused by 30 species), temperate and boreal forests (40 per cent of documented impacts, caused by 28 species), and in cultivated areas (34 per cent of documented impacts, caused by 23 species).

Impacts on the category food and feed, constitute around a quarter of all documented positive impacts in tropical and subtropical dry and humid forests (29 per cent of documented impacts, caused by 35 species), and temperate and boreal forests (24 per cent of documented impacts, caused by 32 species); they also account for almost a fifth of all documented positive impacts in temperate grasslands (18 per cent of documented impacts, caused by 21 species) and cultivated areas (18 per cent of documented impacts, caused by 18 species). Positive impacts on the category, medicinal, biochemical and genetic resources also make a sizeable contribution to documented positive impacts in tundra and high mountain

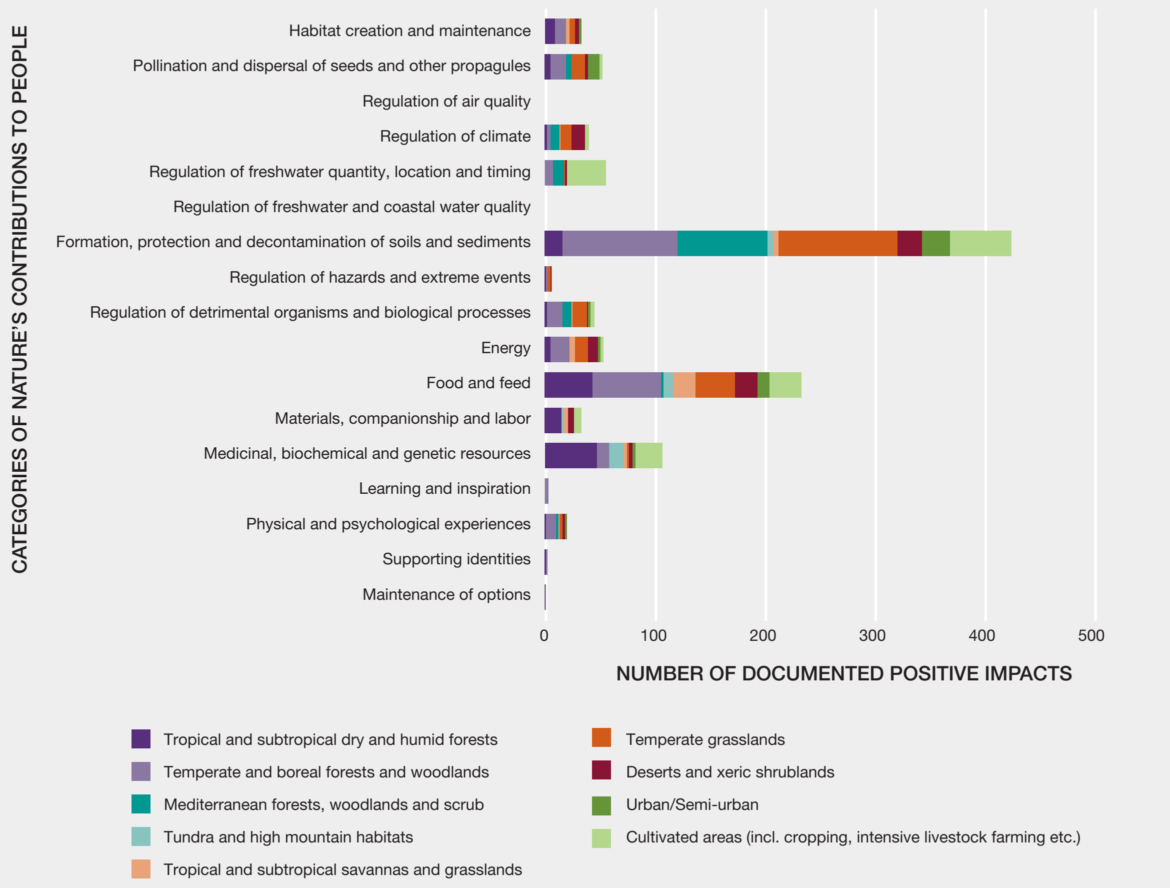


Figure 4 31 **Documented number of positive impacts (x axis) of invasive alien species on categories of nature's contributions to people (y axis) across different terrestrial units of analysis.**

There are 1,103 documented positive impacts across all categories of nature's contributions to people in the terrestrial realm. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

habitats (45 per cent; of documented impacts, caused by 13 species), tropical and subtropical dry and humid forests (32 per cent of documented impacts, caused by 43 species), and cultivated areas (15 per cent of documented impacts, caused by 22 species).

Invasive alien taxa most often documented causing impacts on nature's contributions to people in the terrestrial realm

Plants is generally the invasive alien species taxonomic group that causes the most impacts (both positive and negative) on nature's contributions to people across all units of analysis **(Table 4.15)**. Notable exceptions to this overall pattern occur in cultivated areas and in Mediterranean woodlands, forests and scrub. In cultivated areas, the top 10 species of invasive alien species causing negative impacts are invertebrate crop pests (the top five of which are *Spodoptera frugiperda* (fall armyworm), *Bactrocera dorsalis* (Oriental fruit fly), *Solenopsis invicta* (red imported fire ant), *Chilo partellus* (spotted stem borer), and *Phenacoccus manihoti* (cassava mealybug)). *Spodoptera frugiperda* alone accounts for four to six per cent of maize losses in North and South America and Sub-Saharan Africa (Savary *et al.*, 2019), though yield losses ranging from 10 per cent up to 58 per cent have been estimated across different countries in Africa (**Box 4.18**; **Table 4.26**). Globally, total crop losses (from all pests and pathogens combined) are estimated at 20 to 30 per cent, based on a global survey of crops that together account for about half of human calorie intake (Savary *et al.*, 2019). A large proportion of these total crop losses is due to insect and mite pests, 30 to 45 per cent of which are alien invasive arthropods, as estimated across several large crop-growing countries of the world (Pimentel *et al.*, 2001).

In Mediterranean woodlands, forests and scrub, seven of the top 10 invasive alien species are microbes and include five species of fungi, a bacterium (*Xylella fastidiosa* (Pierce's disease of grapevines)), and an oomycete (*Phytophthora cinnamomi* (Phytophthora dieback)). Negative impacts of these microbes on nature's contributions to people include, more specifically, impacts on habitat creation and maintenance. For example, *Ceratocystis platani* (canker stain of plane), a canker-causing fungus thought to have been accidentally introduced to Europe from North America in the first half of the twentieth century, damages the iconic *Platanus orientalis* (plane), especially in Greece (Tsopelas *et al.*, 2017). Other significant examples of negative impacts are impacts on the provision of food and feed. *Cryphonectria parasitica* (blight of chestnut), which has devastated native North American chestnut populations following its accidental introduction to the United States in the early twentieth century (Anagnostakis, 1987), is also a significant pest in Europe, where it threatens fruit and wood production from the European chestnut (EFSA PLH Panel (EFSA Panel on Plant Health), 2014). Likewise, recently discovered (in 2013) olive quick decline syndrome (OQDS), caused by *Xylella fastidiosa* has caused significant losses to the economically and culturally important olive crop in Italy's main olive growing region (White *et al.*, 2017).

Most invertebrates appearing in the top 10 invasive alien species by unit of analysis are associated with negative impacts on nature's contributions to people. However, exceptions are the positive impacts on pollination, and food and feed by *Apis mellifera* (European honeybee) in temperate and boreal forests, Mediterranean woodland forest and scrub, and cultivated areas. In the United States, for example, the pollination of crops by *Apis mellifera* is in the order of tens of billions of US$ annually (Pejchar & Mooney, 2009). *Apis mellifera* has also been observed to pollinate the culturally significant Hawaiian endemic tree, *Meterosideros polymorpha* (ʽOhi'a), and could play an important role in the future, as species assemblages increasingly change as a result of species invasions, habitat fragmentation, and climate change (Cortina *et al.*, 2019). However, in Latin America, *Apis mellifera* has hybridized with the aggressive Africanized honeybee, and threatens human health (Pejchar & Mooney, 2009). Another invertebrate that was introduced as a pollinator of cultivated plants to many regions of the globe is *Bombus terrestris* (bumble bee). Although it is now an invasive alien species, its use as a pollinator continues, though it is regulated (e.g., in Japan; Goka, 2010). In some cases, invasive alien plants also support pollination providing a reliable source of nectar and pollen at times of the year when agricultural landscapes are otherwise not providing sufficient resources to maintain pollinator populations (Hirsch *et al.*, 2020).

The regulation of detrimental organisms and biological processes is another category of nature's contributions to people that is positively impacted by an invertebrate, e.g., by *Nematus oligospilus* (willow sawfly) in urban/semiurban areas. This species was unintentionally introduced to Australia and New Zealand as a pest of introduced willows (Caron *et al.*, 2014). It is therefore perceived as a beneficial species, since willows are detrimental to riparian and aquatic ecosystems (Bruzzese & McFadyen, 2006).

Some species cause both negative and positive impacts on nature's contributions to people, even within the same unit of analysis. *Prosopis* spp. (mesquite) are an example of this in deserts and xeric shrublands. Positive impacts include the provision of fuelwood, shade, and fodder in the form of pods (S. E. Shackleton & Shackleton, 2018; **Box 4.9**); negative impacts include depletion of groundwater (Dzikiti *et al.*, 2013), a reduction in grazing resources, and damage caused to certain livestock from consumption of pods (Obiri, 2011; **Box 4.9**). *Prosopis juliflora*, *Prosopis pallida*, *Prosopis glandulosa*, *Prosopis chilensis*, and *Prosopis velutina* (collectively known as mesquite) have been introduced across the globe for the potential benefits they provide to people; however, with their spread in introduced regions, the negative impacts of *Prosopis* spp. come into conflict with their positive impacts, creating contradictions in how they are perceived by different stakeholders (R. T. Shackleton *et al.*, 2014; **Box 4.9**).

Some species or taxa cause impacts on nature's contributions to people across multiple units of analysis. One example is the genus, *Acacia*, with different species – *Acacia dealbata* (acacia bernier), *Acacia mearnsii* (black wattle), and *Acacia saligna* (coojong) – causing positive impacts on nature's contributions to people across several different units of analysis. These positive impacts (on formation, protection and decontamination of soils and sediments, climate regulation, energy, and materials, companionship and labour (Lorenzo *et al.*, 2010; Potgieter *et al.*, 2019; C. M. Shackleton *et al.*, 2007) align with a recent global review of the genus *Acacia* and its impacts, which found positive impacts on climate regulation, soil fertility and soil erosion control (Castro-Díez *et al.*, 2021). However, these findings are at odds with other work highlighting the negative impacts of *Acaciae* on various categories of nature's contributions to people, especially regulation of freshwater quantity, location and timing, and regulation of hazards and extreme events; (Le Maitre *et al.*, 2011). Although *Acaciae* are associated with negative impacts on nature's contributions to people in the impact database developed through this chapter[9] as well, their documented negative impacts do not rank amongst the top 10 species by units of analysis.

9. Data management report available at: https://doi.org/10.5281/zenodo.5766069

Table 4 15 **Main invasive alien species impacting nature's contributions to people in the terrestrial realm.**

The top 10 (by number of documented impacts) invasive alien species causing negative and positive impacts on nature's contributions to people in the terrestrial realm by the affected units of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate: Microorganisms:

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxa	Species	Documented impacts	Taxa	Species	Documented impacts
Temperate and boreal forests and woodlands		*Lymantria dispar* (gypsy moth)	26		*Solenopsis invicta* (red imported fire ant)	11
		Solenopsis invicta (red imported fire ant)	25		*Impatiens glandulifera* (Himalayan balsam)	26
		Phytophthora ramorum (sudden oak death)	36		*Solidago gigantea* (giant goldenrod)	15
		Hymenoscyphus fraxineus (ash dieback)	26		*Reynoutria japonica* (Japanese knotweed)	11
		Fusarium circinatum (pitch canker)	17		*Microstegium vimineum* (Nepalese browntop)	6
		Reynoutria spp. (knotweed)	19		*Rubus ulmifolius* (elmleaf blackberry)	6
		Impatiens glandulifera (Himalayan balsam)	18		*Elaeagnus umbellata* (autumn olive)	5
		Reynoutria japonica (Japanese knotweed)	17		*Lonicera japonica* (Japanese honeysuckle)	5
		Solidago gigantea (giant goldenrod)	15		*Impatiens parviflora* (small balsam)	4
		Quercus rubra (northern red oak)	14		*Prunus laurocerasus* (cherry laurel)	4
Cultivated areas (incl. cropping, intensive livestock farming etc.)		*Spodoptera frugiperda* (fall armyworm)	50		*Apis mellifera* (European honeybee)	3
		Bactrocera dorsalis (Oriental fruit fly)	41		*Reynoutria japonica* (Japanese knotweed)	20
		Solenopsis invicta (red imported fire ant)	41		*Solidago gigantea* (giant goldenrod)	10
		Chilo partellus (spotted stem borer)	37		*Centaurea stoebe* (spotted knapweed)	3
		Phenacoccus manihoti (cassava mealybug)	33		*Cortaderia selloana* (pampas grass)	3
		Liriomyza trifolii (American serpentine leafminer)	24		*Cucumis myriocarpus* (gooseberry gourd)	3
		Prostephanus truncatus (larger grain borer)	17		*Phalaris aquatica* (bulbous canarygrass)	3
		Liriomyza sativae (vegetable leaf miner)	12		*Campanula rapunculoides* (creeping bellflower)	2
		Frankliniella occidentalis (western flower thrips)	10		*Cenchrus ciliaris* (buffel grass)	2
		Rastrococcus invadens (fruit tree mealybug)	10		*Columba livia* (pigeons)	10
Deserts and xeric shrublands		*Bromus tectorum* (downy brome)	18		*Carpobrotus* spp. (icelplant)	8
		Cenchrus ciliaris (buffel grass)	14		*Prosopis glandulosa* (honey mesquite)	4
		Eragrostis lehmanniana (Lehmann lovegrass)	5		*Bromus tectorum* (downy brome)	3
		Prosopis spp. (mesquite)	5		*Cirsium arvense* (creeping thistle)	2

Table 4 15

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxa	Species	Documented impacts	Taxa	Species	Documented impacts
Deserts and xeric shrublands		*Erodium cicutarium* (common storksbill)	4		*Erodium cicutarium* (common storksbill)	2
		Tamarix ramosissima (saltcedar)	3		*Prosopis alba* (white carob tree)	2
		Acacia longifolia (golden wattle)	2		*Aerva javanica* (kapok bush)	1
		Hilaria belangeri (curly mesquite)	2		*Carpobrotus acinaciformis* (Eland's sour-fig)	1
		Juniperus osteosperma (Utah juniper)	2		*Casuarina cunninghamiana* (Australian beefwood)	1
		Camelus spp. (camels)	3		*Cenchrus ciliaris* (buffel grass)	1
Tropical and subtropical dry and humid forests		*Lissachatina fulica* (giant African land snail)	36		*Falcataria falcata* (Moluccan albizia)	7
		Laevicaulis alte (tropical leatherleaf slug)	11		*Fraxinus uhdei* (tropical ash)	7
		Solenopsis invicta (red imported fire ant)	7		*Spathodea campanulata* (African tulip tree)	5
		Deroceras reticulatum (grey field slug)	5		*Cinchona pubescens* (quinine tree)	4
		Hypogeococcus spp. (mealybug)	4		*Decalobanthus peltatus* (merremia)	4
		Wasmannia auropunctata (little fire ant)	4		*Salix fragilis* (crack willow)	4
		Phytophthora ramorum (sudden oak death)	5		*Cedrela odorata* (Spanish cedar)	3
		Syzygium jambos (rose apple)	5		*Gleditsia triacanthos* (honey locust)	3
		Ageratum conyzoides (billy goat weed)	4		*Ligustrum lucidum* (broad-leaf privet)	3
		Jasminum fluminense (Brazilian jasmine)	4		*Pteridium aquilinum* (bracken)	3
Temperate Grasslands		*Rosa rugosa* (rugosa rose)	17		*Rosa rugosa* (rugosa rose)	36
		Reynoutria japonica (Japanese knotweed)	12		*Solidago gigantea* (giant goldenrod)	29
		Bromus tectorum (downy brome)	10		*Genista aetnensis* (Mount Etna broom)	11
		Poa pratensis (smooth meadow-grass)	10		*Reynoutria japonica* (Japanese knotweed)	8
		Solidago canadensis (Canadian goldenrod)	10		*Amorpha fruticosa* (false indigo-bush)	7
		Solidago gigantea (giant goldenrod)	10		*Acacia dealbata* (acacia bernier)	6
		Impatiens glandulifera (Himalayan balsam)	9		*Heracleum pubescens* (Sosnowskyi's hogweed)	5
		Senecio inaequidens (South African ragwort)	9		*Impatiens glandulifera* (Himalayan balsam)	5
		Solidago spp. (goldenrod)	9		*Brassica nigra* (black mustard)	4
		Bothriochloa ischaemum (yellow bluestem)	7		*Columba livia* (pigeons)	10

Table 4 15

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxa	Species	Documented impacts	Taxa	Species	Documented impacts
Mediterranean forests, woodlands and scrub		*Xylella fastidiosa* (Pierce's disease of grapevines)	15		*Carpobrotus* spp. (icelplant)	27
		Ceratocystis platani (canker stain of plane)	13		*Acacia dealbata* (acacia bernier)	21
		Seiridium cardinale (cypress canker)	11		*Genista aetnensis* (Mount Etna broom)	6
		Phytophthora cinnamomi (Phytophthora dieback)	10		*Platanus ×hispanica* (London planetree)	6
		Cryphonectria parasitica (blight of chestnut)	9		*Reynoutria ×bohemica* (Bohemian knotweed)	6
		Sphaeropsis sapinea (Sphaeropsis blight)	9		*Elaeagnus umbellate* (autumn olive)	4
		Heterobasidion irregulare (conifer-base polypore)	5		*Impatiens glandulifera* (Himalayan balsam)	4
		Carpobrotus spp. (icelplant)	10		*Lonicera maackii* (Amur honeysuckle)	4
		Lonicera maackii (Amur honeysuckle)	10		*Celastrus orbiculatus* (Asiatic bittersweet)	2
		Arundo donax (giant reed)	5		*Phalaris aquatica* (bulbous canarygrass)	2
Tropical and subtropical savannas and grasslands		*Centaurea solstitialis* (yellow starthistle)	16		*Acacia mearnsii* (black wattle)	2
		Elaeagnus umbellata (autumn olive)	12		*Cenchrus clandestinus* (Kikuyu grass)	1
		Imperata cylindrica (cogon grass)	8		*Cenchrus geniculatus* (spiny burrgrass)	1
		Melia azedarach (chinaberry)	7		*Centaurea solstitialis* (yellow starthistle)	1
		Melinis minutiflora (molasses grass)	7		*Eragrostis curvula* (weeping lovegrass)	1
		Ligustrum sinense (Chinese privet)	6		*Hyparrhenia hirta* (coolatai grass)	1
		Nandina domestica (nandina)	6		*Hyparrhenia rufa* (jaragua grass)	1
		Triadica sebifera (Chinese tallow tree)	6		*Koelreuteria elegans* subsp. *formosana* (flamegold)	1
		Biancaea decapetala (Mysore thorn)	5		*Medicago minima* (small medick)	1
		Holcus lanatus (common velvet grass)	5		*Bubalus bubalis* (Asian water buffalo)	4
Tundra and High Mountain habitats		*Pinus mugo* (mountain pine)	6		*Pinus mugo* (mountain pine)	5
		Bromus tectorum (downy brome)	4		*Eucalyptus globulus* (Tasmanian blue gum)	2
		Bromus inermis (awnless brome)	2		*Artemisia absinthium* (wormwood)	1
		Linaria vulgaris (common toadflax)	2		*Capsella bursa-pastoris* (shepherd's purse)	1
		Melilotus albus (honey clover)	2		*Cenchrus clandestinus* (Kikuyu grass)	1
		Agropyron cristatum (crested wheatgrass)	1		*Erodium cicutarium* (common storksbill)	1
		Bromus hordeaceus (soft brome)	1		*Hypericum perforatum* (St John's wort)	1

Table 4 15

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxa	Species	Documented impacts	Taxa	Species	Documented impacts
Tundra and High Mountain habitats		*Bromus japonicus* (Japanese brome)	1		*Malva neglecta* (common mallow)	1
		Cenchrus clandestinus (Kikuyu grass)	1		*Malva parviflora* (pink cheeseweed)	1
		Erodium cicutarium (common storksbill)	1		*Matricaria chamomilla* (common chamomile)	1
Urban/Semi-urban		*Lissachatina fulica* (giant African land snail)	32		*Trichocorixa verticalis* (water boatman)	2
		Wasmannia auropunctata (little fire ant)	12		*Bombus terrestris* (bumble bee)	1
		Laevicaulis alte (tropical leatherleaf slug)	9		*Nematus oligospilus* (willow sawfly)	1
		Monomorium pharaonis (pharaoh ant)	8		*Lupinus polyphyllus* (garden lupin)	8
		Anoplolepis gracilipes (yellow crazy ant)	7		*Symphyotrichum lanceolatum* (narrow-leaved michaelmas daisy)	7
		Paratrechina fulva (tawny crazy ant)	5		*Impatiens glandulifera* (Himalayan balsam)	4
		Ceratocystis platani (canker stain of plane)	5		*Paspalum distichum* (knotgrass)	2
		Impatiens glandulifera (Himalayan balsam)	12		*Acacia saligna* (coojong)	1
		Lupinus polyphyllus (garden lupin)	7		*Agrostis alba* (redtop)	1
		Heracleum pubescens (Sosnowskyi's hogweed)	6		*Columba livia* (pigeons)	10

4.4.2.2 Patterns of negative and positive impacts of invasive alien species on nature's contributions to people in the inland waters realm

Impacted units of analysis in the inland waters realm

In inland waters, negative impacts of invasive alien species on nature's contributions to people outnumber positive impacts by a ratio of 4:1, with 600 documented negative impacts compared with only 145 documented positive impacts. About 70 per cent of all documented impacts, both negative and positive, on nature's contributions to people in inland waters are from inland surface waters and water bodies/freshwater; the remainder are found to almost equal measure in wetlands, and in aquaculture areas (**Figures 4.32** and **4.33**).

Impacted categories of nature's contributions to people in inland waters

In inland waters, invasive alien species' negative impacts on food and feed predominate, and constitute about 50 per cent of all documented negative impacts on nature's contributions to people in this realm **(Figure 4.32)**. In aquaculture areas, about 90 per cent of negative impacts are on food and feed (caused by 28 species); in inland surface waters/waterbodies 55 per cent of negative impacts are on food and feed (caused by 82 species). Other sizeable negative impacts of invasive alien species on nature's contributions to people are on habitat creation and maintenance in wetlands (44 per cent of documented impacts, 25 species), and on freshwater quantity (10 per cent of documented impacts; 16 species), and water quality (12 per cent; 18 species), in inland surface waters/waterbodies.

Impacts on the category food and feed also account for the majority (60 per cent) of all documented positive impacts in inland waters **(Figure 4.33)**. Impacts on food and feed constitute 95 per cent of all positive impacts in aquaculture areas (caused by 15 species) and about 60 per cent in inland surface waters/waterbodies (caused by 31 species). Positive impacts on the category formation, protection and decontamination of soils and sediments, constitute almost 50 per cent of all documented impacts in wetlands, caused by five species. Other documented positive impacts are on physical and psychological experiences, water quality, and regulation of biological processes (about 10 per cent of documented impacts each), in inland surface waters/ waterbodies.

Invasive alien taxa most often documented causing impacts on nature's contributions to people in the inland waters realm

In aquaculture areas, as well as in inland surface waters/ waterbodies, vertebrates are the invasive alien species causing the most impacts, both positive and negative, on nature's contributions to people **(Table 4.16)**. Species causing positive impacts include *Hypophthalmichthys nobilis* (bighead carp), *Hypophthalmichthys molitrix* (silver carp), *Oreochromis niloticus* (Nile tilapia), *Cirrhinus mrigala* (mrigal carp), several species of catfish such as *Clarias gariepinus* (North African catfish), and also a reptile, *Crocodylus rhombifer* (Cuban crocodile). These species positively impact the provision of food and feed, and reflect the growing contribution of aquaculture to food security, especially in the Asia-Pacific region (De Silva, 2012); fish constitutes the main dietary protein in many countries of the region (e.g., in Cambodia; Nuov *et al.*, 2005). However, many introduced fish species have also become associated with negative impacts on nature's contributions to people **(Box 4.10)**. These include species originally introduced for aquaculture such as *Cyprinus carpio* (common carp), *Oreochromis mossambicus* (Mozambique tilapia), *Oreochromis niloticus*, *Clarias gariepinus*, *Hypophthalmichthys nobili*, *Hypophthalmichthys molitrix*; species originally introduced for aquatic weed control such as *Ctenopharyngodon idella* (grass carp); *Gambusia affinis* (western mosquitofish) introduced to control malaria by feeding on mosquito larvae; and *Poecilia reticulata* (guppy), an ornamental species introduced for the aquarium trade. These species are linked to negative impacts on native fish (an impact on nature; **section 4.3.2**) *via* competition, predation, hybridization, and physical and chemical alteration of habitat (Ciruna *et al.*, 2004), which, in turn, negatively affect the category food and feed. For example, in the River Ganga, India, between 2004 and 2009, fish catch showed a 72 per cent decline of native

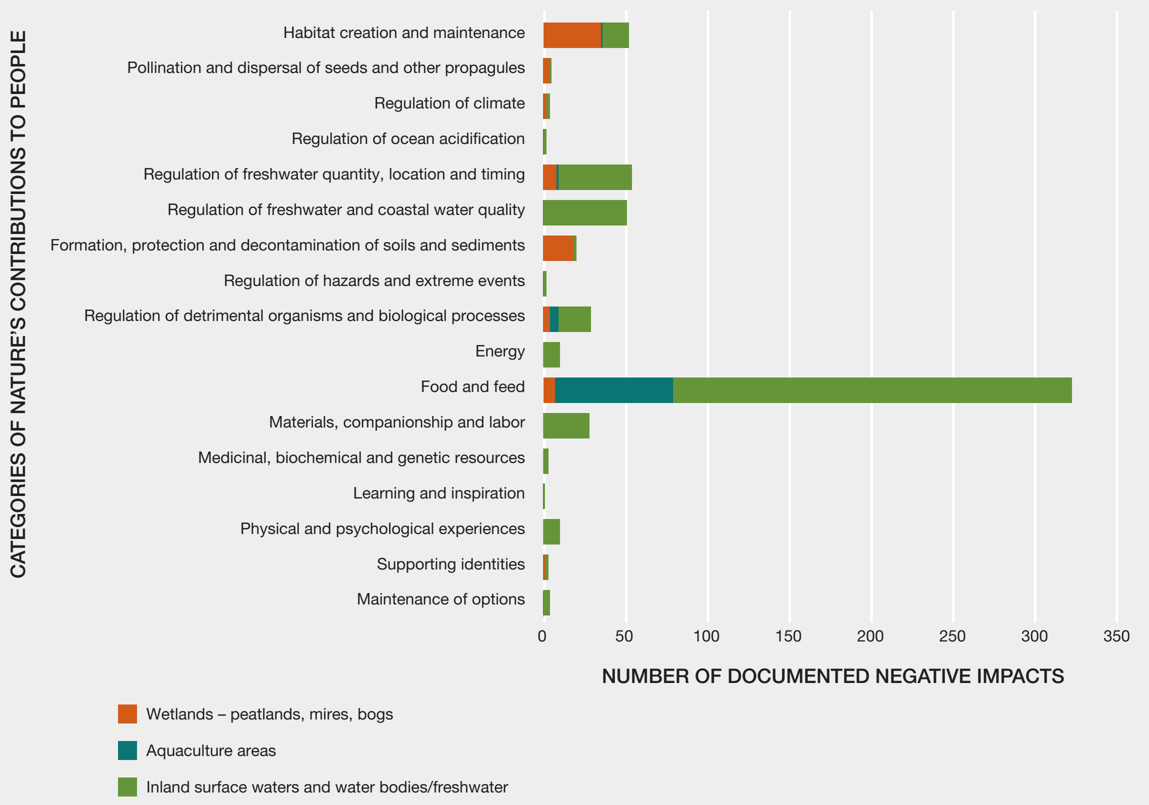


Figure 4 32 **Documented negative impacts (x axis) of invasive alien species on categories of nature's contributions to people (y axis) in the inland waters realm.**

There are 600 documented negative impacts across all categories of nature's contributions to people in the inland waters realm. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

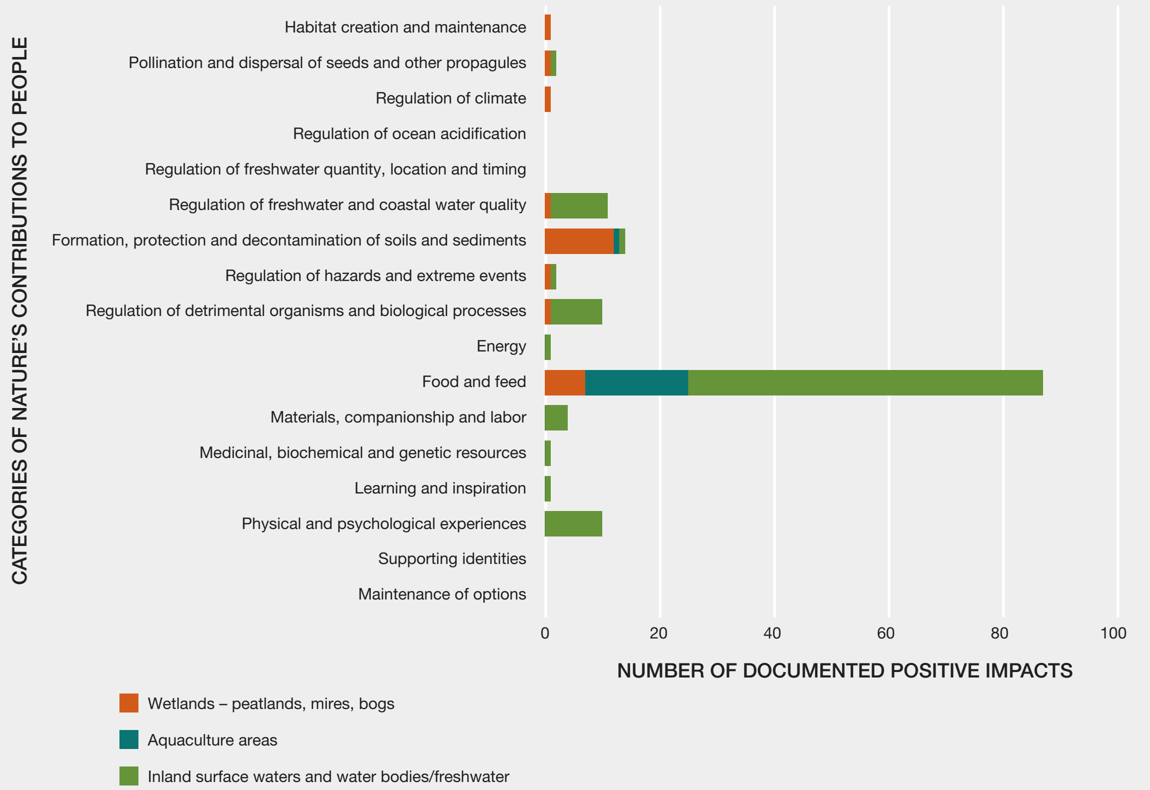


Figure 4 33 **Documented positive impacts (x axis) of invasive alien species on categories of nature's contributions to people (y axis) in the inland waters realm.**

There are 145 documented positive impacts across all categories of nature's contributions to people in the inland waters realm.
A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

fish accompanied by a 237 per cent increase in introduced species (especially *Cyprinus carpio*, *Clarias gariepinus*, and the two species of tilapia (A. K. Singh & Lakra, 2011).

In inland surface waters/waterbodies, the other dominant taxonomic group associated with negative impacts on nature's contributions to people are invertebrates, e.g., *Dreissena polymorpha* (zebra mussel), *Corbicula fluminea* (Asian clam), and *Procambarus clarkii* (red swamp crayfish). Of these, *Dreissena polymorpha* and *Procambarus clarkii* are each responsible for a number of negative impacts on nature's contributions to people. The former affects food and feed, regulation of freshwater and coastal water quality, energy, provision of materials and companionship and labour (Colautti *et al.*, 2006; Strayer, 2009); the latter affects food and feed, habitat creation and maintenance, supporting identities, and learning and inspiration, leading to its evaluation as a high-risk species (Souty-Grosset *et al.*, 2016). Despite its multiple negative impacts, *Procambarus clarkii* also causes positive impacts (on food and feed, and harvested commercially), and live individuals are still bought and sold for the aquarium trade (Souty-Grosset *et al.*, 2016). Other examples of positive impacts by invertebrates are on the categories food and feed by *Pontastachus leptodactylus* (Danube crayfish) (Martinez-Cillero *et al.*, 2019), regulation of detrimental organisms and biological processes by *Gammarus pulex* (common freshwater amphipod) that feeds on mosquito larvae (Dalal *et al.*, 2020), and regulation of freshwater and coastal water quality by *Dreissena rostriformis bugensis* (quagga mussel), which increases water clarity (Verstijnen, 2019).

In wetlands, plants are the invasive alien species causing the most impacts, both negative and positive, on nature's contributions to people, with the exception of two vertebrate species. Negative impacts of plants are largely on the formation, protection and decontamination of soils and sediments. For instance, species such as *Reynoutria japonica* (Japanese knotweed) and *Heracleum pubescens* (Sosnowskyi's hogweed), affect soil food webs by negatively altering nematode communities (Čerevková *et al.*, 2019; Renčo *et al.*, 2019), and *Echinochloa pyramidalis* (limpopo grass) negatively impact habitat creation and maintenance in Mexico (López Rosas *et al.*, 2005). Positive impacts of plants in wetlands include regulation of hazards and extreme events, and regulation of freshwater and coastal water quality. For example, *Phragmites australis* (common reed) in North America can buffer wetlands against the effects of sea level rise, and removes nutrients from agriculture runoff (Kettenring *et al.*, 2012). The vertebrates among the top 10 invasive alien species in wetlands are *Rusa timorensis* (Sunda sambar deer), which has negative impacts on the regulation of freshwater quantity, location and timing due to grazing on vegetation that helps regulate water flows (Pallewatta *et al.*, 2003); and *Bubalus bubalis* (Asian water buffalo), which causes positive impacts on food and feed, according to Indigenous Peoples and local communities in Australia (e.g., Albrecht *et al.*, 2009; C. J. Robinson *et al.*, 2005).

Table 4 16 **Main invasive alien species impacting nature's contributions to people in the inland waters realm.**

The top 10 (by number of documented impacts) invasive alien species causing negative and positive impacts on nature's contributions to people in the inland waters realm by the affected units of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate:

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxon	Species	Documented impacts	Taxon	Species	Documented impacts
Aquaculture areas		*Azolla filiculoides* (water fern)	3		*Azolla filiculoides* (water fern)	1
					Seaweed Cottony II	1
		Cyprinus carpio (common carp)	10		*Hypophthalmichthys nobilis* (bighead carp)	2
		Oreochromis mossambicus (Mozambique tilapia)	9		*Oreochromis niloticus* (Nile tilapia)	2
		Clarias gariepinus (African catfish)	7		*Cirrhinus mrigala* (mrigal carp)	1
		Hypophthalmichthys molitrix (silver carp)	5		*Clarias gariepinus* (North African catfish)	1
		Oreochromis niloticus (Nile tilapia)	5		*Clarias* spp. (catfish)	1
		Ctenopharyngodon idella (grass carp)	4		*Crocodylus rhombifer* (Cuban crocodile)	1
		Gambusia affinis (western mosquitofish)	3		*Gibelion catla* (catla)	1
		Hypophthalmichthys nobilis (bighead carp)	3		*Hypophthalmichthys molitrix* (silver carp)	1
		Poecilia reticulata (guppy)	3			
Inland surface waters and water bodies/ freshwater		*Dreissena polymorpha* (zebra mussel)	36		*Dreissena rostriformis bugensis* (quagga mussel)	6
		Corbicula fluminea (Asian clam)	9		*Dreissena polymorpha* (zebra mussel)	3
		Procambarus clarkii (red swamp crayfish)	9		*Procambarus clarkii* (red swamp crayfish)	3
		Alternanthera philoxeroides (alligator weed)	9		*Gammarus pulex* (common freshwater amphipod)	2
		Cyprinus carpio (common carp)	20		*Pontastacus leptodactylus* (Danube crayfish)	2
		Oreochromis niloticus (Nile tilapia)	15		*Pistia stratiotes* (water lettuce)	4
		Oreochromis mossambicus (Mozambique tilapia)	13		*Oreochromis niloticus* (Nile tilapia)	5
		Hypophthalmichthys molitrix (silver carp)	12		*Bubalus bubalis* (Asian water buffalo)	4
		Clarias gariepinus (North African catfish)	11		*Lates niloticus* (Nile perch)	4
		Ctenopharyngodon Idella (grass carp)	9		*Gambusia affinis* (western mosquitofish)	3

Table 4 16

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxon	Species	Documented impacts	Taxon	Species	Documented impacts
Wetlands – peatlands, mires, bogs		*Echinochloa pyramidalis* (limpopo grass)	6		*Trichocorixa verticalis* (water boatman)	2
		Reynoutria japonica (Japanese knotweed)	5		*Sporobolus pumilus* (saltmeadow cordgrass)	8
		Heracleum pubescens (Sosnowskyi's hogweed)	5		*Heracleum pubescens* (Sosnowskyi's hogweed)	2
		Frangula alnus (alder buckthorn)	4		*Phragmites australis* (common reed)	2
		Elymus athericus (wildrye)	3		*Cenchrus clandestinus* (Kikuyu grass)	1
		Phalaris arundinacea (reed canary grass)	3		*Elymus athericus* (wildrye)	1
		Phragmites australis (common reed)	3		*Reynoutria japonica* (Japanese knotweed)	1
		Sporobolus pumilus (saltmeadow cordgrass)	3		*Impatiens glandulifera* (Himalayan balsam)	1
		Lythrum salicaria (purple loosestrife)	2		*Typha domingensis* (southern cattail)	1
		Rusa timorensis (Sunda sambar deer)	4		*Bubalus bubalis* (Asian water buffalo)	3

4.4.2.3 Patterns of negative and positive impacts of invasive alien species on nature's contributions to people in the marine realm

Impacted units of analysis in the marine realm

In the marine realm, as with the terrestrial and inland waters realms, negative impacts of invasive alien species outweigh positive impacts on nature's contributions to people (85 and 24 documented impacts, respectively). Documented impacts of invasive alien species on nature's contributions to people are predominantly from shelf ecosystems (about 70 per cent of negative and over 85 per cent of documented positive impacts). The remainder of invasive alien species impacts are documented from coastal areas; there are no documented impacts from other marine units of analysis. There are far fewer documented impacts of invasive alien species impacts on nature's contributions to people from marine compared with terrestrial and inland waters units of analysis, which might largely be due to a bias in research efforts.

Impacted categories of nature's contributions to people in the marine realm

Negative impacts on the provision of food and feed predominate in shelf ecosystems and constitute 85 per cent of all documented impacts (caused by 23 species; **Figure 4.34**). Negative impacts in coastal areas are distributed across all categories of nature's contributions to people, though food and feed accounts for a large proportion of documented impacts (approximately 45 per cent, caused by 4 species); other categories of negative impacts in coastal areas include regulation of freshwater and coastal water quality (25 per cent documented impacts; 3 species), maintenance of options (17 per cent of documented impacts, 4 species), and materials, companionship and labour (12 per cent of documented impacts, 2 species).

Positive impacts (as with negative impacts) on the provision of food and feed predominate in both marine units of analysis **(Figure 4.35)**. In the case of shelf ecosystems, food and feed constitutes 76 per cent of all documented impacts (caused by 16 species), with impacts on medicinal, biochemical and genetic resources accounting for an additional 14 per cent of documented impacts (caused by 1 species). Positive impacts of invasive alien species in coastal areas are all on the provision of food and feed (100 per cent of the three documented impacts, caused by 3 species).

This predominance of invasive alien species impacts (both negative and positive) on food and feed in the marine realm documented in the chapter impacts database matches findings from a recent European review of marine invasive

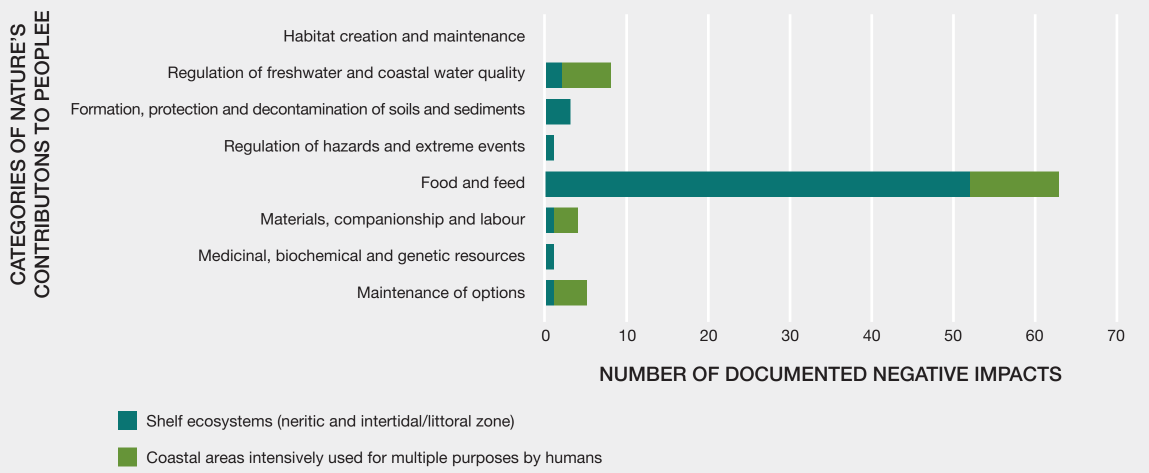


Figure 4 34 **Documented negative impacts (x axis) of invasive alien species on categories of nature's contributions to people (y axis) across different marine units of analysis.**

There are 85 documented negative impacts across all categories of nature's contributions to people in the marine realm. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

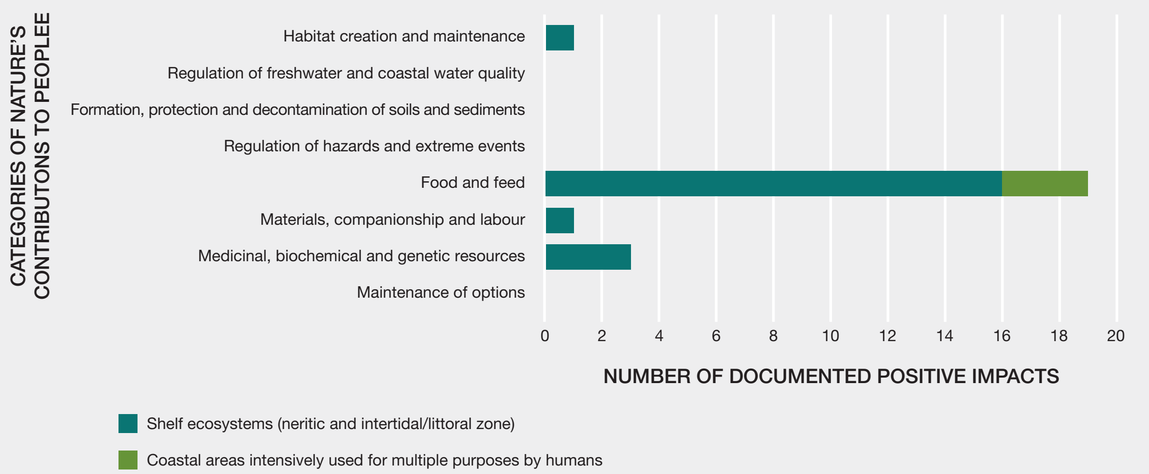


Figure 4 35 **Documented positive impacts (x axis) of invasive alien species on categories of nature's contributions to people (y axis) across different marine units of analysis.**

There are 24 documented positive impacts across all categories of nature's contributions to people in the marine realm. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

alien species and their impacts on marine ecosystem services (Katsanevakis *et al.*, 2014). Of all ecosystem services derived from marine ecosystems, Katsanevakis *et al.* (2014) found the highest number of documented invasive alien species with negative and positive impacts to be on food provisioning (fisheries, aquaculture); however, as suggested by the authors, this could reflect a study bias towards impacts on food, given its societal and economic relevance over other ecosystem services from marine ecosystems.

Invasive alien taxa most often documented causing impacts on nature's contributions to people in the marine realm

In coastal areas, the most documented invasive alien species causing both negative and positive impacts on nature's contributions to people are all invertebrates **(Table 4.17)**. The most documented negative impacts are on food and feed. Examples include impacts of *Carcinus maenas* (European green crab), which feeds on native oysters and crabs, and has decimated commercial shellfish beds in New England and Canada (Pimentel *et al.*, 2000), and the generalist predators, *Asterias amurensis* (northern Pacific seastar) and *Ciona intestinalis* (sea vase), which affect mariculture and fisheries along the Korean coast (Seo & Lee, 2009). Another example is *Mytilopsis sallei* (Caribbean false mussel), which competitively displaces native clams and oysters that are locally important fishery resources in India (Kumar, 2019). In coastal areas, the documented positive impacts on nature's contributions to people caused by invasive alien species are all related to food and feed. All three species, *Magallana gigas* (Pacific oyster), *Penaeus vannamei* (whiteleg shrimp), and *Ruditapes philippinarum* (Japanese carpet shell), are commercially harvested (A. N. Cohen & Carlton, 1995; U.S. Congress, Office of Technology Assessment, 1993).

In shelf ecosystems, invasive alien invertebrates are responsible for the majority of both negative and positive impacts on nature's contributions to people, mostly on food and feed. For example, *Carcinus maenas* (European green crab), through predation, and *Styela clava* (Asian tunicate), through competition for space, have led to a reduction in populations of native species in fisheries (Colautti *et al.*, 2006). Positive impacts of invertebrates are, likewise, associated with the provision of food and feed. For example, *Rapana venosa* (veined rapana whelk) in Turkey (Aydin *et al.*, 2016) and *Penaeus aztecus* (northern brown shrimp) in the Nile Delta of Egypt (Sadek *et al.*, 2018), are both harvested commercially. In addition to the documented invertebrate species, there are two plant species that feature in the list of top 10 invasive alien species: *Codium fragile* (dead man's fingers) and *Gracilaria vermiculophylla* (red alga). *Codium*

Table 4 17 **Main invasive alien species impacting nature's contributions to people in the marine realm.**

The top 10 (by number of documented impacts) invasive alien species causing negative and positive impacts on nature's contributions to people in the inland waters realm, by the affected units of analysis. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate:

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxa	Species	Documented impacts	Taxa	Species	Documented impacts
Coastal areas intensively used for multiple purposes by humans		*Carcinus maenas* (European shore crab)	6		*Magallana gigas* (Pacific oyster)	1
		Mytilopsis sallei (Caribbean false mussel)	4		*Penaeus vannamei* (whiteleg shrimp)	1
		Asterias amurensis (northern Pacific seastar)	3		*Ruditapes philippinarum* (Japanese carpet shell)	1
		Ciona intestinalis (sea vase)	2			
		Teredo navalis (naval shipworm)	2			
		Batillaria attramentaria (Japanese false cerith)	1			
		Magallana gigas (Pacific oyster)	1			
		Littorina littorea (common periwinkle)	1			
		Lyrodus pedicellatus (blacktip shipworm)	1			
		Mytella strigata (Charru mussel)	1			

Table 4 17

Unit of analysis	Invasive alien species with negative impacts on nature's contributions to people			Invasive alien species with positive impacts on nature's contributions to people		
	Taxa	Species	Documented impacts	Taxa	Species	Documented impacts
Shelf ecosystems (neritic and intertidal/littoral zone)		*Carcinus maenas* (European shore crab)	7		*Penaeus aztecus* (northern brown shrimp)	4
		Mytella strigata (Charru mussel)	5		*Laguncula pulchella* (predatory sea snail)	3
		Ficopomatus enigmaticus (tubeworm)	3		*Mytella strigata* (Charru mussel)	2
		Styela clava (Asia tunicate)	3		*Mytilus galloprovincialis* (Mediterranean mussel)	2
		Ascidiella aspersa (European sea squirt)	2		*Rapana venosa* (veined whelk)	2
		Ciona intestinalis (sea vase)	2		*Cercopagis pengoi* (fishhook waterflea)	1
		Ciona robusta (tunicate)	2		*Paralithodes camtschaticus* (red king crab)	1
		Tubastraea spp. (sun corals)	2		*Gracilaria vermiculophylla* (red alga)	1
		Codium fragile (dead man's fingers)	14		*Pterois volitans* (red lionfish)	2
		Pterois volitans (red lionfish)	2		*Megalops atlanticus* (Atlantic tarpon)	1

fragile has been associated with losses to commercial eel, lobster, and oyster fisheries in Canada (Colautti *et al.*, 2006). *Gracilaria vermiculophylla* is harvested for extraction of agar, which is used in the food industry (A. M. M. Sousa *et al.*, 2010). There are also two vertebrate species on this top 10 list, *Pterois volitans* (red lionfish) and *Megalops atlanticus* (Atlantic tarpon). *Pterois volitans*, a predator, has caused population declines of native fish on which local fisherfolk depend (Miguez Ruiz, 2013). Analogous to the much better documented introductions through the Suez canal that is regarded as an extremely significant route of introduction for marine invasions (Galil *et al.*, 2015), *Megalops atlanticus* is thought to have arrived in the eastern Pacific through the Panama Canal and was initially documented in the 1940s. It now extends on approximately 2600 km along the Pacific coastline of Central and South America (Castellanos-Galindo *et al.*, 2019). Its impacts are perceived as positive by different communities along the Colombian coast as it is used as a resource for food, crafts, and also for game fishing in the tourism industry (Neira & Acero P, 2016).

4.4.3 Documented impacts on nature's contributions to people by region and taxonomic group

Many invasive alien species are causing negative impacts on nature's contributions to people in all regions, with documented impacts from more than 500 species in the Americas and Europe and Central Asia (538 and 531, respectively), followed by 314 species in the Asia-Pacific region and 136 in Africa **(Figure 4.36)**. Across regions, some of these invasive alien species also have positive impacts on nature's contributions to people, and the percentage of species with documented positive impacts ranges from 22 per cent of species in Europe and Central Asia, 26 per cent in Africa, 32 per cent in the Americas, to 41 per cent in the Asia-Pacific region.

Most impacted categories of nature's contributions to people by region, and by taxon

Across all regions, food provisioning is the most impacted category of nature's contributions to people, both negatively and positively. Negative impacts on provisioning of food are found in all regions and for all taxa **(Figure 4.37)**. In Africa, most invasive alien species causing these impacts are plants (59 species), followed by invertebrates (36) and vertebrates (22), and with just one documented microbe species (Maize lethal necrosis disease). A similar pattern is observed for the Americas (131 plants, 76 invertebrates, 30 vertebrates, 22 microbes), whereas in the Asia-Pacific and Europe and Central Asia regions, invasive alien invertebrates are the largest taxonomic group causing impacts on food provisioning (81 and 217, respectively). The highest number of invasive alien vertebrates (74 species) causing impacts on food provisioning is found in the Asia-Pacific region.

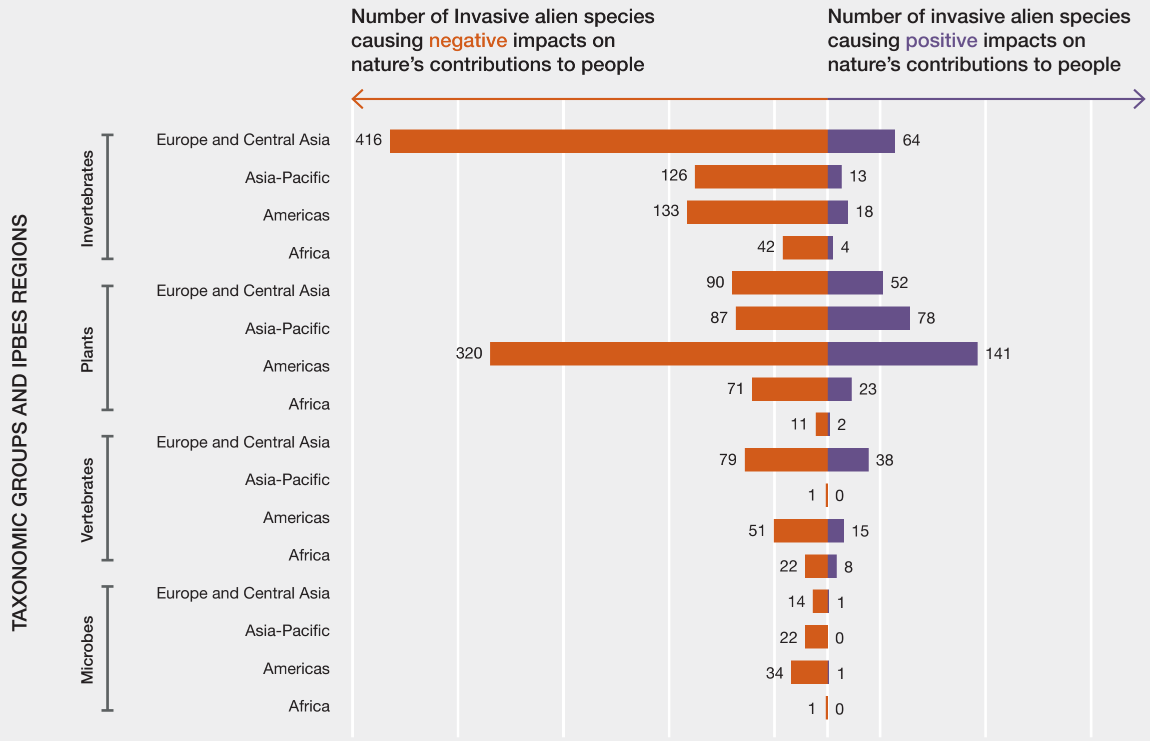


Figure 4 36 **Number of invasive alien species with documented negative and positive impacts on nature's contributions to people (x axis) per region and taxonomic group (y axis).**

Positive and negative stacked bar charts do not imply that positive and negative impacts can be summed. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

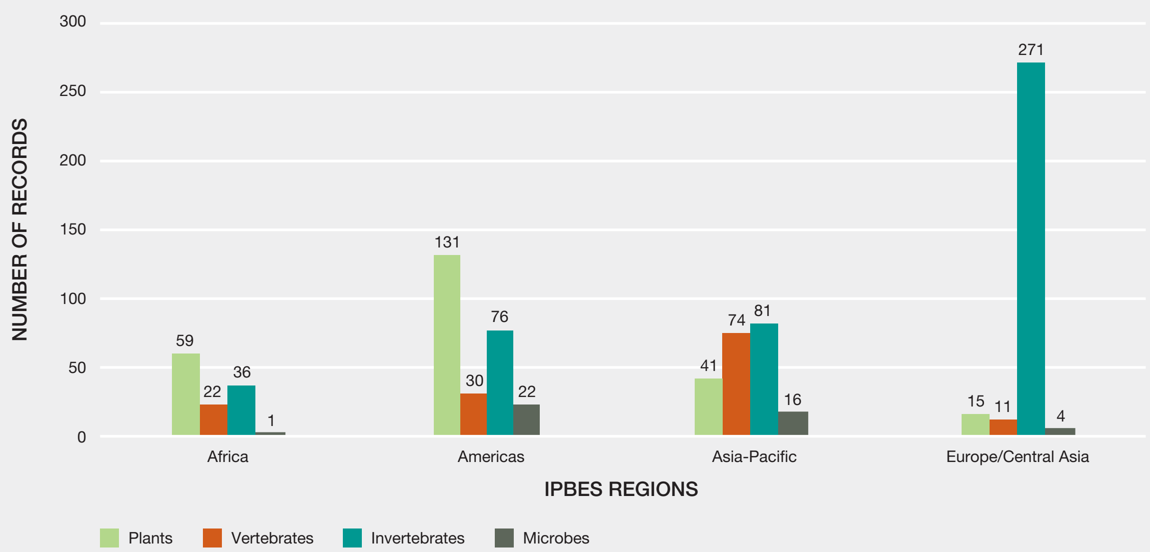


Figure 4 37 **Number of documented impacts (y axis) of invasive alien species on food provisioning by region (x axis).**

The number of invasive alien species involved is indicated above each column. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Invasive alien taxa most often documented causing negative impacts by region

There are more impacts caused by plants in Africa, the Americas and the Asia-Pacific regions than in Europe and Central Asia **(Figure 4.37)**. In comparison to other regions, rangelands used for grazing livestock are less common in Europe and Central Asia than other regions of the world (Boone *et al.*, 2018), where high numbers of invasive alien species impact food provisioning through overgrowing rangelands or by harming livestock with poisonous or injurious parts **(Box 4.9)**. Impacts caused by invasive alien plants (as weeds in agricultural crops) are often not distinguished from impacts caused by native plants, especially when the impacts of the entire weed flora are assessed (Vilà *et al.*, 2004), and are therefore likely to be underrepresented in the impact database developed through this chapter.[10]

10. Data management report available at: https://doi.org/10.5281/zenodo.5766069

Pimentel *et al.* (2005) have estimated the impacts of invasive alien weeds in the United States by using the percentage of invasive alien agricultural weed species in the total weed flora to proportionate the crop losses caused by alien weeds. As a result, they estimated crop losses of US$24 billion annually based on the assumption of 12 per cent crop losses caused by weeds, of which 73 per cent were allocated to invasive alien weeds, corresponding to their share in the US agricultural weed flora.

A more comprehensive analysis of the impacts of plants across the four regions **(Table 4.18)** shows that in Europe and Central Asia, the highest number of different categories of nature's contributions to people is affected. In particular, impacts on soils by plants have been documented to be caused by 45 invasive alien species in this region, but there are also high numbers of plants impacting negatively on pollination (19) and biological processes (17). Across all regions, freshwater provision is the category of

Table 4 18 **Number of invasive alien species with negative impacts on nature's contributions to people by taxonomic group and region.**

Acronyms used in the table: NCP – nature's contributions to people; Af – Africa; Am – Americas; AP – Asia-Pacific; ECA – Europe and Central Asia. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

NCP	PLANTS				VERTEBRATES				INVERTEBRATES				MICROBES			
	Af	Am	AP	ECA	Af	Am	AP	ECA	Af	Am	AP	ECA	Af	Am	AP	ECA
Habitat	7	174	34	1	0	3	1	0	0	23	1	4	0	2	2	13
Pollination	1	46	0	19	0	1	0	0	0	5	5	2	0	0	0	0
AirQuality	0	2	8	1	0	0	0	0	0	1	0	0	0	0	0	0
Climate	0	4	0	6	0	0	1	0	0	2	0	0	0	0	0	0
OceanAcid	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Freshwater	17	21	14	12	0	1	1	0	0	2	0	6	0	0	0	0
WaterQuality	12	9	6	2	0	3	3	0	0	4	3	3	0	0	0	0
Soils	6	22	5	45	0	4	2	0	0	3	0	1	0	1	0	0
Hazards	1	25	7	3	0	2	1	0	0	2	0	2	0	0	0	0
BiolProcess	3	25	5	17	0	3	4	0	6	26	13	4	0	0	0	0
Energy	4	4	2	0	0	1	1	0	0	4	0	0	0	0	0	0
Food	59	131	41	15	22	30	74	11	36	76	81	217	1	22	16	4
Materials	8	15	13	0	0	5	4	0	1	37	39	173	0	10	4	0
Medicinal	4	3	4	0	0	0	0	0	0	3	1	5	0	0	0	1
Learning	0	1	1	0	0	0	0	0	0	1	0	1	0	0	0	0
Physical	1	3	3	24	0	4	1	0	1	13	12	13	0	0	2	0
Identities	0	1	8	0	0	0	0	0	0	0	0	2	0	0	1	0
Options	10	0	4	0	0	34	0	0	0	28	2	0	0	0	1	0

Table 4.19 **Number of invasive alien species with positive impacts on nature's contributions to people by taxonomic group and region.**

Acronyms used in the table: Af – Africa, Am – Americas, AP – Asia-Pacific, Ant – Antarctica, ECA – Europe and Central Asia, NCP – nature's contributions to people. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

	PLANTS				VERTEBRATES				INVERTEBRATES				MICROBES			
NCP	Af	Am	AP	ECA	Af	Am	AP	ECA	Af	Am	AP	ECA	Af	Am	AP	ECA
Habitat	0	21	6	0	0	0	1	0	0	1	0	0	0	0	0	1
Pollination	0	5	0	11	0	1	1	0	0	1	3	0	0	0	0	0
AirQuality	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Climate	9	0	2	13	0	0	0	0	0	0	0	0	0	0	0	0
OceanAcid	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Freshwater	0	18	3	5	0	0	0	0	0	0	0	0	0	0	0	0
WaterQuality	0	3	1	1	0	0	0	0	0	0	0	3	0	0	0	0
Soils	0	11	26	38	0	0	0	0	0	6	0	0	0	0	0	0
Hazards	0	3	3	4	0	0	0	0	0	0	0	0	0	0	0	0
BiolProcess	2	8	1	10	0	3	2	0	0	5	2	6	0	0	0	0
Energy	12	5	6	2	0	0	0	0	0	0	0	0	0	0	0	0
Food	13	66	21	7	5	11	34	2	3	10	7	41	0	1	0	0
Materials	5	14	14	2	0	0	0	0	2	2	0	10	0	0	0	0
Medicinal	6	42	33	4	0	0	0	0	0	0	2	1	0	0	0	0
Learning	0	0	0	0	0	2	0	0	0	0	0	2	0	0	0	0
Physical	0	0	3	6	5	1	1	0	0	3	0	1	0	0	0	0
Identities	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0
Options	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0

nature's contributions to people that is more consistently negatively impacted by invasive alien plants. The number of documented impacts on material provisions by invasive alien invertebrates is very high in Europe and Central Asia (173 invasive alien species) and is also high in the Americas (37) and Asia-Pacific (39) regions, but there is only one record in Africa. In Africa in particular, impacts on forestry have not been as well documented (both in the literature and in the impact database developed through this chapter)[11] as, for example, in North America (e.g., Aukema *et al.*, 2011) or Europe and Central Asia. Europe also has a higher rate of documented new introductions of forest pests and diseases than other continents (Kenis *et al.*, 2017; Santini *et al.*, 2013).

Among immaterial nature's contributions to people, impacts on the maintenance of options by invasive alien plants have been documented in Africa (10) and Asia-Pacific (4), by vertebrates (34) and invertebrates (28) in the Americas, and for one microbe species in Asia-Pacific. Physical and psychological experiences have been impacted by 24 invasive alien plants in Europe and Central Asia, but also by invertebrates in the Americas, Asia-Pacific and Europe and Central Asia regions. There are eight plants with impacts on the "supporting identities" category, a further single record of one plant in this category for the Americas, of two invertebrates for Europe and Central Asia and of one microbe from the Asia-Pacific region.

Invasive alien taxa most often documented causing positive impacts by region

Many invasive alien species (mostly plants) also provide benefits to people, which, in many cases, have been the reason for their initial or continued introduction **(Table 4.19)**. Food provisioning can be improved by invasive alien plants

11. Data management report available at: https://doi.org/10.5281/zenodo.5766069

and vertebrates in all regions. In the Americas, 66 invasive alien plants have documented uses as food or feed, but numbers are lower in other regions, with 21 documented impacts in Asia-Pacific, 13 in Africa and 7 in Europe and Central Asia. Invasive alien vertebrate species are mostly used for food and feed in the Asia-Pacific region (34 species), whereas in the Europe and Central Asia region, invertebrates (41 species) are the largest invasive alien species taxa group providing this category of nature's contributions to people. Further benefits from plants are documented for soils in Europe and Central Asia (38 species) and Asia-Pacific (26 species). Invasive alien plants are used across all regions for medicinal reasons and for energy generation. Benefits for climate, for example through carbon sequestration in soils, have been documented for invasive alien plants in Europe and Central Asia (13 species), in Africa (9 species), mainly for micro-climate impacts of shade and windbreaks and in Asia-Pacific (2 species). Invasive alien plants also provide material benefits (e.g., as timber), and 14 species have been documented for this category of nature's contributions to people both in Asia-Pacific and the Americas, with fewer species in the other regions. Most documented impacts of the use of invasive alien plants for energy are from Africa, where 12 woody invasive alien species are documented as being used as firewood.

4.5 IMPACTS OF INVASIVE ALIEN SPECIES ON GOOD QUALITY OF LIFE

4.5.1 General patterns

Many of all documented impacts of invasive alien species are known to directly or indirectly affect good quality of life (15.7 per cent, 3,783 impacts), ranging from impacts on people's material and immaterial assets (e.g., food, housing), health (**section 4.5.1.3**), safety, relationships with people and nature, and maintaining opportunities for the future (i.e., the different constituents of good quality of life; **section 4.1.1**, **Box 4.3**). Globally, 1,032 invasive alien species have documented impacts on good quality of life, with 900 invasive alien species causing negative impacts and 236 causing positive impacts. Among those, 104 invasive alien species cause both positive and negative impacts, with both benefits and costs for good quality of life. These particular species can pose challenges for decision makers because they are differently perceived by different stakeholders (**Chapter 5**, **section 5.6.1.2**; **Figure 4.38**). The 796 invasive alien species causing only negative impacts is reflective of the higher number of negative impacts documented for good quality

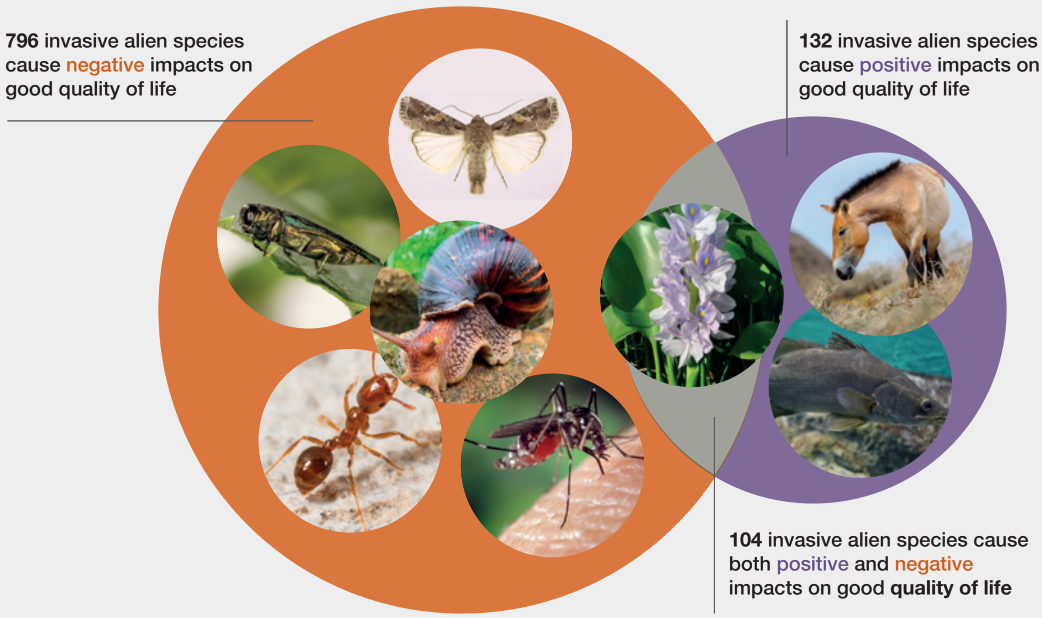


Figure 4 38 **Invasive alien species mostly cause negative impacts on good quality of life.**

Number of invasive alien species causing only negative impacts on good quality of life (left), negative and positive impacts (centre) and positive impacts only (right). Please note that the size of the segments is not proportional to the numbers. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069. Photo credits: Stephen Ausmus, USDA Agricultural Research Service – Public domain (Emerald ash borer) / Marián Polák, WM Commons – CC BY-SA 4.0 (wild horse) / Daiju Azuma, WM Commons – CC BY-SA 4.0 (Nile perch) / Bharat B. Shrestha – CC BY 4.0 (water hyacinth) / Canadian Biodiversity Information Facility, WM Commons – Public domain (fall armyworm) / Sonel, pixabay – CC BY 4.0 (Giant African land snail) / James Gathany, CDC, WM Commons – Public domain (Asian tiger mosquitos) / elharo, Adobe Stock – Copyright (red imported fire ant).

of life, which is spread across all taxa, regions and units of analysis (**sections 4.5.2** and **4.5.3**).

Most impacted constituents of good quality of life

Globally, the most impacted constituents of good quality of life, both in terms of negative and positive impacts, are material and immaterial assets **(Table 4.20)**, which account for almost two-thirds (60.4 per cent, 2286 impacts) of the documented impacts. The number of negative documented impacts is approximately six times higher than that of the positive impacts, which is likely due to the fact that this constituent is directly related to livelihood of people, and is therefore documented more frequently. Positive impacts on material and immaterial assets have often resulted from the initial introduction of the invasive alien species for a specific purpose, for example food crops or plants delivering materials for fuel, which help people to improve their quality of life. However, if these species spread into natural areas, they can lead to negative impacts such as reduction in food gathered from nature, or yields from forestry and fisheries, therefore contributing to economic hardship, poverty and food insecurity for the same people or different stakeholders. Alternatively, invasive alien plants may cause initial negative impacts on material and immaterial assets, then people can adapt and find some benefits from the species. For example, after the intentional introduction of the tree *Prosopis juliflora* (mesquite) as forage for livestock and habitat stabilization, which led to widespread loss of native grassland, people have adapted to novel *Prosopis*-based livelihoods, especially by making charcoal and harvesting the wood for sale **(Box 4.9)**; this livelihood diversification and increased financial capital has enabled communities to cope better with environmental shocks (Sato, 2013; Walter & Armstrong, 2014). Parts of the tree have traditionally been used for medicinal purposes, and people are adapting it for medicinal use in its introduced habitats as well (Damasceno *et al.*, 2017; Duenn *et al.*, 2017). It is important to note that some communities do not voluntarily adapt to a species and its positive impact; they may not have had a choice, and their preferred option may still be the native species. Furthermore, adaptation does not necessarily increase the resilience (**Glossary**) of socioecological systems (**section 4.6**). Although some invasive alien species may be considered "useful" by particular groups of stakeholders, their presence is likely to have negative consequences for others, creating potential for conflict. In the Eastern Cape of South Africa, for example, *Opuntia ficus-indica* (prickly pear) provides a source of food and income for some local communities, but negatively impacts subsistence farmers by reducing the carrying capacity of land for livestock. The capacity to derive benefits such as food or energy from *Opuntia* spp. can even vary within local communities, whereby some women's groups are able to produce biogas and *Opuntia* jam and fruit juice, while others in the community do not have this capacity (IPBES, 2022).

Invasive alien species also greatly affect human health, which accounts for nearly one quarter of the impacts on good quality of life (22.2 per cent, 839 documented impacts), with 87 per cent of those impacts being negative (**Table 4.20**, **Boxes 4.15** and **4.16**). Together, the documented impacts on social, spiritual and cultural relations, safety and freedom of choice and action represent 14.2 per cent (536 documented impacts) of all the impacts on good quality of life and are also mainly negative impacts (77.8 per cent, **Table 4.20**). Many invasive alien species reduce access to grazing areas and water sources, resulting in food insecurity

Table 4 20 **Number of negative and positive impacts on constituents of good quality of life caused by invasive alien species.**

The number of impacts documented for each constituent of good quality of life. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Constituent of good quality of life	Negative impacts	Positive impacts
Material and immaterial assets	2005	281
Health	728	111
Social, spiritual and cultural relations	240	97
Safety	82	18
Freedom of choice and action	95	4
Unknown	58	64
Grand Total (%)	3208 (85%)	575 (15%)

and possible conflicts among pastoralist communities **(Box 4.9)**. The cultivation of *Acacia mangium* (brown salwood) has been documented to threaten the cultural and material continuity of the Wapichana and Macuxi in Brazil (Souza *et al.*, 2018). Invasive alien plants that form dense monocultures in semi-arid ecosystems, such as *Cenchrus ciliaris* (buffel grass), can physically block access to culturally important places, reducing socially valuable native species, and ultimately changing the opportunities for cultural knowledge transmission and well-being of future generations (Read *et al.*, 2020). Physical damage to ecosystems (e.g., changing water quality), can have a negative impact on people for whom water sources are sacred and central to their culture and well-being **(Box 4.14)**. Human safety is directly at risk from, for instance, falling branches due to dead or dying trees as a result of invasive alien species outbreaks, and indirectly impacted from, for instance, increased intensity of fires caused by more flammable invasive grasses (**Chapter 1**, **Box 1.4**). Impacts of invasive animals on personal safety are however more concerning, with, for instance, *Sturnus vulgaris* (common starling) nests near airports that put human lives and aviation equipment at risk regularly (Linz *et al.*, 2007). Larger invasive animals such as *Sus scrofa* (feral pig) or *Camelus dromedarius* (dromedary camel) are also known to directly scare or attack people or cause collisions and road accidents (Koichi *et al.*, 2012; Vaarzon-Morel, 2010). Invasive alien plants indirectly reduce people's safety as larger shrubs and trees have been documented to harbour wildlife that encroach on human settlements, increasing human-wildlife conflicts and impacting the safety of people; this has been especially documented by Indigenous Peoples and local communities (Puri, 2015; Sundaram *et al.*, 2012).

Ratio of positive and negative impacts on good quality of life

More than 6 out of 7 (3208 impacts, 85 per cent) documented impacts of invasive alien species on good quality of life are negative, and far fewer (575 impacts, 15 per cent) are positive for good quality of life. The ratio of negative to positive impacts on good quality of life caused by invasive alien species, is approximately 6 to 1. Other reviews of impacts of invasive alien species on human livelihoods have found higher proportions of beneficial impacts being documented. For instance, R. T. Shackleton, Shackleton, *et al.* (2019) have reviewed 51 case studies, in which 86 per cent of case studies documented detrimental impacts on human livelihoods and 79 per cent documented positive impacts on livelihoods. Similarly, in a review of 70 case studies, P. L. Howard (2019) has found that 90 percent of the case studies examined show evidence of harmful impacts on various ecosystem services and livelihood measures, while approximately 65 percent of the case studies document at least some positive effects. These reviews generally include groups of people with high dependence on nature for livelihoods, including, but not limited to, Indigenous Peoples and local communities. With a close proximity and reliance on natural resources, such communities may be the first to experience impacts of invasive alien species, but they also may be able to adapt and derive benefits when livelihoods are at stake (P. L. Howard, 2019; **section 4.6**). Therefore, the comparison of costs and benefits of invasive alien species will vary depending on the social-economic context, and some do not consider benefits as wholly "positive", and instead form part of trade-offs or more complex perspectives (IPBES, 2022).

Gender-differentiated impacts

Indigenous Peoples and local communities, ethnic minorities, migrants, poor rural and urban communities are disproportionately impacted by invasive alien vector-borne diseases (**section 4.5.1.4**; Bardosh *et al.*, 2017; Molyneux *et al.*, 2011). Gender bias is documented in some studies of invasive alien species impact upon people's livelihoods. Gender-differential impacts occur when invasive alien species limit or provide a resource which is gender-preferentially utilized. Male-dominated artisanal fisheries in Lake Victoria, dependent on tilapia for livelihood and food security, has declined due to the invasion of *Pontederia crassipes* (water hyacinth). The impact of *Pontederia crassipes* on the catchability was more important in the Kenyan section of Lake Victoria, where the tilapia population was reduced by 45 per cent (Kateregga & Sterner, 2009; Ongore *et al.*, 2018). In the gender-based division of labour among Rabari pastoralists in northwest India, men are responsible for the herd's health, access to water sources, fodder, camping sites, and face the negative impacts of *Prosopis juliflora* (mesquite; Duenn *et al.*, 2017). Similarly, women among the buffer zone community forest users of Chitwan National Park, Nepal, who are responsible for collecting grasses and fodder, reported that the invasive *Mikania micrantha* (bitter vine) makes collection of forest resources increasingly difficult (Rai & Scarborough, 2015; Sullivan *et al.*, 2017). The invasion of the forest reserve in Chamarajanagar, Karnataka, India, by *Lantana camara* (lantana) has been perceived by the neighbouring Lingayat women as contributing to the decline of a native palm. Palm-leaf broom making is one of the few income earning options available to them in the village (Kent & Dorward, 2015).

Some invasive alien species, such as *Acacia mearnsii* (black wattle), are widely used by Indigenous Peoples and local communities. In the Eastern Cape, South Africa, rural communities made widespread consumptive use, with 97 per cent of households in rural communities collecting wattle for fuelwood and building or fencing. While 53 per cent of the community members (men & women) prefer high densities of the shrub, 10 per cent fear criminals hiding in the *Acacia mearnsii* forests (C. M. Shackleton *et al.*, 2007).

Some invasive alien species provide food, fuel and income to women belonging to Indigenous Peoples and local communities, and help bring them into the mainstream economic activity. In most developing countries, the majority of marginalized coastal villagers impacted by invasive alien seaweed (e.g., *Kappaphycus, Eucheuma*) farming and small-scale processing are women. Economic gains from seaweed farming contributed to positive changes in the quality of life, in food, shelter, clothing, health care and social acceptance (Krishnan & Kumar, 2010; Msuya & Hurtado, 2017; Rameshkumar & Rajaram, 2019). In South Sulawesi, Indonesia, women documented seaweed farming as generating 50 per cent or more of their household income (Larson *et al.*, 2021; Rimmer *et al.*, 2021). In Kibuyuni, on the south coast of Kenya, women comprise 75 per cent of seaweed farmers. The income earned has empowered them to participate in societal issues and family decision-making processes (Mirera *et al.*, 2020).

There is very little systematic research on gender differences in impacts of invasive alien species beyond anecdotal evidence of direct impacts (for further examples see IPBES, 2022). The available data suggest that invasive alien species may, on occasion, cause impacts that are gender-biased, and gender-differentiated positive impacts may, in some cases, outweigh negative ones.

Invasive alien species most often documented causing negative impacts on good quality of life

The impact database developed through this chapter shows that there is a subset of invasive alien species that cause a disproportionate negative impact on good quality of life.[12] One-quarter of all negative impacts on good quality of life are caused by only 3 per cent (29 species) of all invasive alien species **(Table 4.21)**.

Six of the top 10 invasive alien species with the highest frequency of negative impacts on good quality of life **(Table 4.21)** are the same as those in the top 10 list of those invasive alien species that cause negative impacts on nature's contributions to people in the food and feed category (**section 4.4.1**; **Table 4.14**), which demonstrates

12 Data management report available at: https://doi.org/10.5281/zenodo.5766069

Table 4 21 **Main invasive alien species causing negative impacts on good quality of life.**

The top 10 (by number of documented impacts) invasive alien species causing negative impacts on good quality of life, organized by highest frequency of documented impacts. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate: Microorganisms:

Invasive alien species	Taxa	Frequency of negative impacts documented for constituents of good quality of life					
		Assets	Health	Relations	Safety	Freedom	Total
Lissachatina fulica (giant African land snail)		42	40	0	0	0	82
Dengue virus		30	38	0	0	8	76
Solenopsis invicta (red imported fire ant)		32	39	0	3	0	74
Pontederia crassipes (water hyacinth)		0	27	6	0	18	51
Spodoptera frugiperda (fall armyworm)		46	0	0	0	0	46
Bactrocera dorsalis (Oriental fruit fly)		40	0	0	0	0	40
Phenacoccus manihoti (cassava mealybug)		35	0	0	0	0	35
Phytophthora ramorum (sudden oak death)		32	0	0	0	0	32
Hymenoscyphus fraxineus (ash dieback)		26	0	0	0	0	26
Cyprinus carpio (common carp)		24	0	0	0	0	24

how invasive alien species can harm good quality of life by negatively impacting the quality and availability of services and contributions from nature. *Spodoptera frugiperda* (fall armyworm; **Box 4.18**), *Bactrocera dorsalis* (Oriental fruit-fly), *Phenacoccus manihoti* (cassava mealybug), and *Lissachatina fulica* (giant African land snail) are serious pests of crops that affect the nature's contributions to people food and feed category, which then flows onto to affect people's access to material assets and support their livelihoods. Impacts of invasive alien species may also flow to other foundations underpinning good quality of life such as human health, for example, not only does *Lissachatina fulica* cause damage to crops, with Indigenous Peoples and local communities reporting that farmers have had to abandon their farms in Antigua and Barbuda after this damage, but they also affect human health by carrying a parasite that causes meningitis (IPBES, 2020).

In addition to serious crop pests, the top 10 most frequently documented invasive alien species with negative impacts on good quality of life **(Table 4.21)** include two microbial pathogens that cause dieback of plants, in this case important tree species such as *Fraxinus* spp. (ash) and *Quercus* spp. (oak), which are valued worldwide for amenity, climate regulation, and cultural traditions (Poland *et al.*, 2017). As a group, microbes causing tree dieback impact multiple constituents of good quality of life. For example, *Phytophthora cinnamoni* (Phytophthora dieback), *Phytophthora agathidicida* (kauri dieback), and *Austropuccinia psidii* (myrtle rust), which causes dieback of myrtaceous species such as the cultural keystone tree *Agathis australis* (kauri) in New Zealand, and *Eucalyptus* and *Melaleuca* (paperbarks) globally. *Eucalyptus* and *Melaleuca* support multiple livelihoods and major industries: where *Austropuccinia psidii* has caused a decline in tree health and subsequent yield losses of up to 70 per cent in *Melaleuca* oil plantations, fungicides needed to be applied, which limited the freedom of choice and action for growers, as they no longer had the option to be certified as organic producers (Carnegie & Pegg, 2018). Kauri trees are a key part of ancestral stories and spirituality for Māori in New Zealand (referred to as a taonga species), and the observed dieback of mature kauri trees in New Zealand has caused widespread concern about the potential impacts on spiritual

Table 4 22 **Main invasive alien species causing negative impacts on more than one constituent of good quality of life.**

The top 10 (by number of documented impacts) invasive alien species causing negative impacts on more than one constituent of good quality of life, representing invasive alien species with the broadest impacts on good quality of life. Dark shading represents species affecting 3 constituents, light shading those affecting 2 constituents. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate: Microorganisms:

Invasive alien species	Taxa	Constituents of good quality of life					
		Assets	Health	Relations	Safety	Freedom	Number of constituents affected
Dengue virus		Yes	Yes			Yes	3
Solenopsis invicta (red imported fire ant)		Yes	Yes		Yes		3
Pontederia crassipes (water hyacinth)			Yes	Yes		Yes	3
Dreissena polymorpha (zebra mussel)			Yes	Yes		Yes	3
Agrilus planipennis (emerald ash borer)				Yes	Yes	Yes	3
Lissachatina fulica (giant African land snail)		Yes	Yes				2
Wasmannia auropunctata (little fire ant)			Yes	Yes			2
Sus scrofa (feral pig)				Yes	Yes		2
Ailanthus altissima (tree-of-heaven)				Yes	Yes	Yes	2
Lymantria dispar (gypsy moth)				Yes		Yes	2

and cultural relationships, although more collaboration is needed between Māori knowledge and science knowledge systems to document how impacts on nature are also affecting good quality of life (Kauri Protection Governance Group, 2022).

Table 4.22 presents the top 10 invasive alien species causing negative impacts on more than one constituent of good quality of life, with a different ranking to the one based solely on the highest number of documented impacts for any category **(Table 4.21)**. It represents a broader range of impacts beyond material and immaterial assets. Some species appear on both tables: for example, Dengue virus causes damage to material and immaterial assets and health as well as freedom of choice and action and has a high number of reports across these categories. Other invasive viruses and human disease-causing microbes may well rank highly for impacts on good quality of life but have not been as well incorporated in the impact database developed through this chapter.[13] Some invasive alien species, whilst not documented as frequently as those in **Table 4.22**, were documented on a broader range of constituents of good quality of life, including *Dreissena polymorpha* (zebra mussel), *Lymantria dispar* (gypsy moth), *Ailanthus altissima* (tree-of-heaven), and *Agrilus planipennis* (emerald ash borer) **(Box 4.14)**. *Sus scrofa* (feral pig) affects human social and cultural relations and safety, and is probably representative of other invasive hard-hooved larger herbivores, whereby

13. Data management report available at: https://doi.org/10.5281/zenodo.5766069

Box 4 14 Impacts of emerald ash borer on Kanienkehá:ka (Mohawk) and W8banaki (Abénakis) Nations lands and the interaction with proposed policy responses.

Agrilus planipennis (emerald ash borer) is an invasive beetle from Asia whose lifecycle is dependent on ash trees. This invasive alien species was first discovered near the Great Lakes region of North America in 2002 and has since spread widely, killing millions of ash trees in North America (Haack *et al.*, 2002; Herms & McCullough, 2014). Many Indigenous nations have a special relationship to the ash tree, especially *Fraxinus nigra* (black ash – Maahlakws in Aln8ba8dwaw8gan (w8banaki language) and éhsa in Kanien'kéha (Mohawk language). Black ash is used in traditional arts such as basketry (Frey *et al.*, 2019; Poland *et al.*, 2017). In the past and still today, the loss of access to black ash due to land privatization, environmental pressures, and the emerald ash borer has had a significant impact on basket making (Blanchet *et al.*, 2022). In turn this results in a loss of traditional knowledge and language about this important cultural practice.

More than handicraft, basketry represents a symbol of cultural resilience for many nations, like the Kanienkehá:ka (Mohawk) and W8banaki (Abénakis) **(Figure 4.39)**. The practice survived despite all odds and the many obstacles that colonization and governmental restrictions have imposed over centuries. The art of basket making is embedded in Kanien'kehá:ka and W8banaki culture, identity, and spirituality. It has also been an important source of income for generations and continues today. According to the W8banakiak creation story, they come from black ash and their existence has always been interwoven. According to oral tradition, without the tree, W8banakiak (the -ak marks the plural) would not exist and if it would disappear, the Nation would as well. Furthermore, still today, many funeral urns are made of black ash. Thus, their identity "stems from the species", according to Martin Gill, Aln8ba from Odanak, (Blanchet *et al.*, 2022), and black ash is what holds the community together.

In 2018, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessed the black ash tree as threatened (COSEWIC, 2018). This assessment prompted the Canadian government to consider listing the species under the federal *Species at Risk Act* (SARA). Listing the black ash in Canadian law would provide funding for recovery, but it would also impose restrictions such as a ban on selling ash baskets, and this would have devastating impacts on indigenous basketry, even though the practice of indigenous basketry is not causing the decline of black ash – the emerald ash borer is causing this decline. This approach also creates conflict between the Canadian view of conservation and Indigenous rights and relationship to the black ash. As an alternative approach, Indigenous Peoples have been actively researching and implementing measures to assure the protection and conservation of the black ash species, including monitoring, treatment, seed saving, and research collaborations (Poland *et al.*, 2017; Reo *et al.*, 2017) including W8banaki collaborations with University of Laval, Quebec. For example, sharing Indigenous and local knowledge, in this case about submerging black ash logs prior to harvest, has since been transferred to mainstream management as a suitable technique to reduce the survival of emerald ash borer (Poland *et al.*, 2017).

Banning the sale of baskets would significantly impact the ability to practice basketry, especially considering the existing pressures on the practice. This would then be a direct negative impact on the autonomy, rights and cultural identity of the Indigenous Peoples who practice this tradition and is counter-productive to the principal of self-determination.

"This would be the beginning of the end for basketry, teachings must continue so that the practice can be transmitted to future generations. We can't survive off love [and good faith] alone – we can't be giving them [baskets] away or keeping them for ourselves" Daniel G. Nolett, Aln8ba from Odanak (Blanchet *et al.*, 2022).

Box 4 14

"They would be taking a part of me. I cannot begin to conceive my existence without my relationship to black ash" Suzie O'Bomsawin, Aln8baskwa from Odanak (Blanchet *et al.*, 2022).

Women are likely to be particularly affected if there were a ban on the sale of baskets. For women, the sale of baskets is an incentive for not only their production and importance in household economies, but also for the intergenerational transmission of associated knowledge and skills. Long term impacts of this approach could see any entire generation lose access to basketry skills, and basketry practice and associated cultural identity would likely disappear.

Figure 4 39 **Basketry, a symbol of cultural resilience for the Kanienkehá:ka (Mohawk) and W8banaki (Abénakis).**

Black Ash trees and the process of basketmaking using prepared strips of Black Ash wood (right) has special cultural significance for many First Nations in North Amercia. Baskets (centre and left) are more than handicrafts, as they are symbols of cultural identity and resilience, the practice supports knowledge transfer between generations and they are integral to local economies, especially for women. Emerald Ash Borer has killed millions of Black Ash trees and policies to protect the Black Ash, such as banning the sale of baskets, may further threaten the cultural, social and economic livelihoods of First Nations people. Photo credit: Musée des Abénakis – Copyright.

they damage important cultural sites and species, and can attack people or cause road accidents (C. J. Robinson & Wallington, 2012; Vaarzon-Morel, 2010).

Invasive alien species most often documented causing positive impacts on good quality of life

In contrast to the top 10 invasive alien species causing negative impacts on good quality of life, which are mainly invertebrates and microbes, positive impacts on good quality of life are being derived primarily from invasive alien plants and vertebrates **(Table 4.23)**. These invasive alien plants have been introduced to multiple countries to either provide land and/or water rehabilitation, ornamental or shade purposes (*Robinia pseudoacacia* (black locust), *Pontederia crassipes* (water hyacinth), *Ailanthus altissima* (tree-of-heaven)), or livelihood resources (*Proposis juliflora* (mesquite)), from which people in multiple countries derive benefits that improve their quality of life (**section 4.6**). However, all of these four species in particular have negative impacts on nature, nature's contributions to people and good quality of life as well, with *Ailanthus altissima* also listed as one of the top 10 invasive alien species with negative impacts on more than one aspect of well-being. These four invasive alien plant species have been well-studied in the literature and thus, benefits to good quality of life are frequently documented alongside negative impacts.

Similarly, the three vertebrates in this top 10 listing of positive impacts also have negative impacts on nature, nature's contributions to people and good quality of life. *Cyprinus carpio* (common carp), *Oreochromis niloticus* (Nile tilapia), and *Oreochromis mossambicus* (Mozambique tilapia) are freshwater fish that have been introduced and adapted to as a food source and to support fishing industries and livelihoods, from which people derive material assets **(Box 4.10)**, although some of these adaptations may not have been the preferred option for some local communities (**section 4.6**). *Cyprius carpio* is also listed as a top 10 species causing negative impacts on good quality of life **(Table 4.21)**.

Three other invasive alien species that provide benefits to people are *Equus ferus* (wild horse), *Eucheuma denticulatum* (eucheuma seaweed), and *Columba livia* (pigeons). Wild horses are considered by some groups of people as a culturally-important species and thus benefit social and cultural relations (Collin, 2017), *Eucheuma denticulatum* contains medicinal properties, and *Columba livia* is used as a food source by local communities.

Table 4 23 **Main invasive alien species causing positive impacts on good quality of life.**

The top 10 (by number of documented impacts) invasive alien species causing positive impacts on good quality of life. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

Plants: Vertebrate:

Invasive alien species	Taxa	Frequency of positive impacts documented for constituents of good quality of life					
		Assets	Health	Relations	Safety	Freedom	Total
Robinia pseudoacacia (black locust)	Plants	30	0	8	8	0	46
Prosopis juliflora (mesquite)	Plants	10	0	0	4	0	14
Pontederia crassipes (water hyacinth)	Plants	7	6	0	1	0	14
Columba livia (pigeons)	Vertebrate	10	0	0	0	0	10
Equus ferus (wild horse)	Vertebrate	0	0	6	0	1	7
Cyprinus carpio (common carp)	Vertebrate	6	0	0	0	0	6
Eucheuma denticulatum (eucheuma seaweed)	Plants	6	0	0	0	0	6
Oreochromis mossambicus (Mozambique tilapia)	Vertebrate	6	0	0	0	0	6
Oreochromis niloticus (Nile tilapia)	Vertebrate	6	0	0	0	0	6
Ailanthus altissima (tree-of-heaven)	Plants	4	0	2	0	0	6

4.5.1.1 Invasive alien ants impact multiple constituents of good quality of life

Invasive alien ants are a group of invasive alien species with a high number of impacts documented across multiple constituents of good quality of life, particularly affecting human health, more so than material assets. Invasive ants have been well-studied in terms of socio-economic impacts (Gruber *et al.*, 2022). Using the SEICAT **(Box 4.2)**, 550 socio-economic impacts of invasive ants have been documented for 65 named species in 50 countries and territories, with most documented impacts from the United States (36 per cent), Brazil (22 per cent), Australia (5 per cent) and Malaysia (5 per cent). The most frequently identified socio-economic impacts are on health (60.6 per cent of documented impacts) and material assets (35.1 per cent). The remaining impacts are on social (4.7 per cent), spiritual (0.4 per cent), and cultural relations (2.4 per cent) and non-material assets (1.9 per cent). Health impacts (269/279 documented impacts) are predominantly from stings and bites, and some deaths have also been documented.

Vectoring of pathogens in hospitals and food preparation facilities have been considered minor health impacts. Effects on material assets are mostly electrical damage from ants nesting in appliances and infrastructure, and damage to crops and livestock that affected livelihoods, also usually to a minor degree (128 out of 153 documented impacts). Impacts on non-material assets and social, spiritual, and cultural relations include avoidance of outdoor activities and health effects on pets. Under the SEICAT methodology, which categorizes a species based on the highest magnitude of documented impact, *Wasmannia auropunctata* (little fire ant) poses the most serious socio-economic threat (massive impact as determined by permanent disappearance of an activity), followed by *Solenopsis invicta* (red imported fire ant) and *Anoplolepis gracilipes* (yellow crazy ant). However, these highest categorizations were each based on a single record only. Of these three species, *Anoplolepis gracilipes* is the most widespread having a pan-tropical distribution. The introduced range of *Wasmannia auropunctata* is predominantly in the Caribbean, but also includes some islands in the Pacific, eastern Australia, western Africa, southern United States, and in Europe and northern America. *Solenopsis invicta* has been introduced to the southern United States, the Caribbean, China, Japan, and Australia. All other species are ranked as having, at most, moderate impacts (changes in activity size, switching or moving activities) or minor impacts (difficult to carry out normal activities).

4.5.1.2 Small island states and the impact of invasive alien species on good quality of life

One fifth of the documented impacts (20.5 per cent, 776 impacts) of invasive alien species on good quality of life are found in island states. Among these, 76.5 per cent (594) are negative impacts, and 23.5 per cent (182) are positive impacts **(Figure 4.40)**. While the proportion of negative to positive impact cases in the islands is about 3.3 (594/182), this figure for the mainlands is about 6.7 (2614/394), suggesting positive impacts on good quality of life are proportionally higher in islands than on mainland. These numbers suggest that island ecosystems and good quality of life in islands are vulnerable to invasive alien species and their impacts (D'Antonio & Dudley, 1995; **section 4.3.1.1**). Both positive and negative impacts are primarily documented on material and immaterial assets, such as food production in agriculture, followed by health and relations. About half of the positive impacts of good quality of life in health and relations (52 of 111 cases, and 52 of 97 cases, respectively) are documented from island states. Islands tend to lack some of the species that can be beneficial for human uses, and thus inhabitants introduced alien species to improve their good quality of life. This could explain why positive impacts on good quality of life are documented at a higher rate from islands. For the same reason, agriculture introductions are the major pathway of invasive alien species introduction in island states (Driscoll *et al.*, 2014). This probably leads to the proportionally larger number of documented cases in cultivated lands. Documented impact across units of analysis in islands are disproportionally distributed: while the number of documented negative impact cases from dry land in islands is equal to the mainlands (201 and 214), there are only 4 cases from boreal forest as opposed to 292 cases in mainlands. This is probably because studied islands are predominantly biased to the tropics in the Asia-Pacific region. Indeed, 70.2 per cent of the documented impacts on good quality of life in islands are found in Asia-Pacific, although Asia-Pacific represents 40.9 per cent of all documented impacts on good quality of life. Invertebrates have caused 55.4 per cent of negative and 46.7 per cent positive impacts on the good quality of life in islands, while this figure is 50 per cent and 28.9 per cent in mainlands. While plants are the largest group with positive impacts in mainlands, invertebrates are the largest group with positive impacts in islands.

Figure 4 40 **Number of impacts (y axis) of invasive alien species on constituents of good quality of life (x axis) in islands and in mainlands.**

This figure shows the number of negative (-) and positive (+) impacts on good quality of life in islands and in other locations globally, and for each constituent of good quality of life. A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

In islands, two aquatic plant species, *Pontederia crassipes* (water hyacinth) and *Salvinia × molesta* (kariba weed), are the major species that reduce good quality of life through compromising assets and health constituents, followed by *Lissachatina fulica* (giant African land snail), Dengue virus, and *Laevicaulis alte* (tropical leatherleaf slug). Among invertebrates, molluscs are the major invasive alien species groups that provide positive impacts on good quality of life in islands. *Lissachatina fulica* and *Laevicaulis alte* are the two most documented invasive alien species from islands causing both positive and negative impacts. They are major agriculture pests (Cowie *et al.*, 2009) as well as intermediate hosts for various parasites, such as *Angiostrongylus cantonensis* (rat lungworms; Barratt *et al.*, 2016). At the same time, those molluscs were documented to positively impact assets and health through predating invasive alien molluscs or transmitting parasites to their final hosts, such as invasive alien rats (Nurinsiyah & Hausdorf, 2019). Studies on the positive impacts of invasive alien species on good quality of life are still limited, and information on the positive impacts caused by molluscs is based on a single study (Nurinsiyah & Hausdorf, 2019).

4.5.1.3 Direct and indirect impacts on human health

Invasive alien species negatively impact human health, from nuisance to allergies, poisoning, disease and death (Martinou & Roy, 2018; **Figure 4.41**; **Box 4.15**).

The widely dispersed agricultural and garden pest *Lissachatina fulica* (giant African land snail; **Figure 4.42**) serves as the intermediate host of a parasitic nematode, *Angiostrongylus cantonensis* (rat lungworm). Human infection with its larvae, through handling or consumption, causes eosinophilic meningitis, which may result in cranial nerve abnormalities, ataxia, encephalitis, coma, and, rarely, death (Kwon *et al.*, 2013; Malvy *et al.*, 2008; Thiengo *et al.*, 2010; Tsai *et al.*, 2001). The aggressive and venomous invasive alien ant *Solenopsis invicta* (red imported fire ant; **Figure 4.42**), introduced from Brazil to the southern United States, Caribbean, East Asia, and Australia, represents a significant health hazard. Its venom induces an immediate, severe burning sensation; subsequent reactions may range from local pustules and rash to life-threatening anaphylaxis (deShazo *et al.*, 1990, 1999; deShazo & Banks, 1994; Stafford, 1996; Xu *et al.*, 2012; Zhang *et al.*, 2007). Incidence of fatalities attributed to fire ant-induced anaphylaxis have been rare to none in the southeastern United States (Prahlow & Barnard, 1998; Rhoades *et al.*, 1989). Between 30 and 60 per cent of the population in urban areas infested by imported fire ants are stung every year (deShazo *et al.*, 1990).

Many vector-borne pathogens have appeared in the past few decades in new regions as result of introductions, some causing explosive epidemics (Kilpatrick & Randolph, 2012; **Box 4.16**). Zoonotic diseases transmitted by invasive mosquito genera (e.g., *Aedes*, *Anopheles*, *Culex*) include

Box 4 15 Direct and indirect impacts of invasive alien species on human health.

Invasive alien species occasionally have deleterious impacts on human health, presenting serious challenges to the good quality of life (**Chapter 1**, **section 1.6.7.2**). They can affect physical as well as psychological health (Martinou & Roy, 2018), directly (e.g., injury to people) or indirectly (e.g., through a reduction in food security). Their role as disease vectors is discussed in **Box 4.16**.

In the terrestrial realm, biological invasions are directly affecting people. There are many invasive alien terrestrial plants with highly allergenic pollen, including *Ambrosia artemisiifolia* (common ragweed), native to Central and Northern America but now found throughout the world, and the dermatitis-causing *Heracleum mantegazzianum* (giant hogweed), native to southern Russia and Georgia but now spread through northern Europe (Jakubska-Busse *et al.*, 2013; Klimaszyk *et al.*, 2014; Lim *et al.*, 2021). *Solenopsis invicta* (red imported fire ant), invasive in North America since the 1930s, inflicts severe stings and has killed people with allergies to its venom (Jemal & Hugh-Jones, 1993). Invasive alien snakes were introduced in Guatemalan oil palm plantations to limit rodent populations, and have bitten local children living nearby, forcing families to relocate (IPBES, 2020). Invasive agricultural pests can also indirectly affect human health by reducing food security; for example, the income and nutrition of small holder farmers and their families involved in mixed maize farming in east Africa is hampered by several major invasive alien species, including *Chilo partellus* (spotted stem borer) and viruses causing Maize Lethal Necrosis Disease (C. F. Pratt *et al.*, 2017). *Spodoptera frugiperda* (fall armyworm) has been described as an "emerging food security global threat" by the Food and Agricultural Organization of the United Nations (FAO) and International Plant Protection Convention; its impact has been painfully evident in countries facing other severe challenges to public health and governance, such as the Democratic Republic of the Congo, Sudan, and Yemen (FAO, 2018). Food security is tightly linked to invasive alien species management in China (McBeath & McBeath, 2010) and wheat-producing countries such as the United States and Canada need to protect against a variety of pernicious invasive alien species such as *Trogoderma granarium* (khapra beetle), one of the world's worst storage pests (Athanassiou *et al.*, 2019).

In inland waters, the shells of *Dreissena polymorpha* (zebra mussel) can cause skin injuries to recreational swimmers and commercial fishers. *Pontederia crassipes* (water hyacinth)

Box 4 15

can make small-scale freshwater fishing next to impossible, indirectly lowering income, food security, and nutrition levels for local communities. Moreover, its introduction has been implicated in the spread of malaria in Lake Victoria due to the creation of habitat for the mosquitoes that harbour *Plasmodium* parasites (Kasulo & Perrings, 2000).

In the marine realm, venomous and poisonous invasive alien species include *Plotosus lineatus* (striped eel catfish) and *Lagocephalus sceleratus* (silver-cheeked toadfish), urchins and jellyfish (Galanidi *et al.*, 2018; Galil, 2018; **Figure 4.41**).

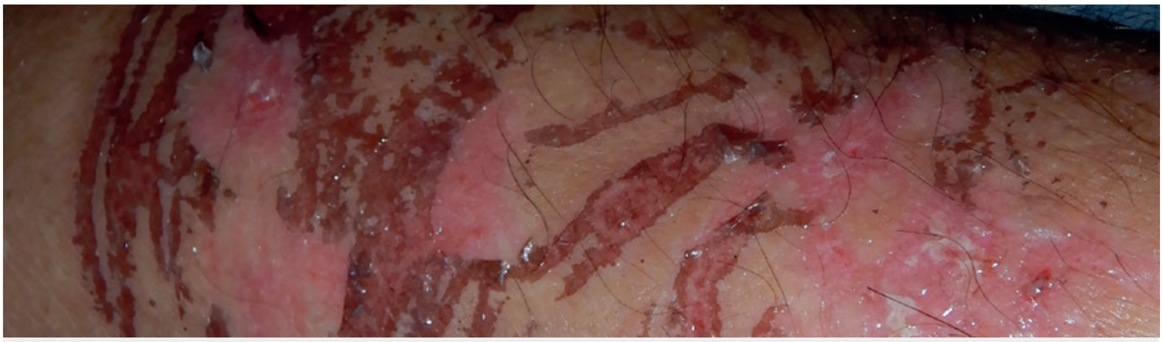


Figure 4 41 **Injuries inflicted by the invasive jellyfish *Rhopilema nomadica* (nomad jellyfish).**

Rhopilema nomadica are invasive alien species found in the Mediterranean Sea. Their stings impact good quality of life in this area. Photo credit: Moti Mendelson – CC BY 4.0.

Combined with other threats to good quality of life described elsewhere in this assessment, invasive alien species directly and indirectly present formidable challenges to human health, in the midst of climate change (Schindler *et al.*, 2018). Awareness of the extent of the threat posed to human health by invasive alien species is still limited. Studies on the impacts of invasive alien species on mental health impacts are also emerging. For example, a participant in an IPBES Indigenous and local knowledge workshop and a formal study both suggest there has been a noticeable decrease in "subjective well-being" due to the impacts of the invasive *Agrilus planipennis* (emerald ash borer) in North America (IPBES, 2020; B. A. Jones, 2017; **Box 4.18**).

malaria, dengue fever, chikungunya, Zika, yellow fever, and West Nile fever, and inflict misery, chronic symptoms, and occasionally death (M. R. Duffy *et al.*, 2009; Effler *et al.*, 2005; Enserink, 2006; Fares *et al.*, 2015; Heukelbach *et al.*, 2016; Kilpatrick, 2011; Laras *et al.*, 2005; Nash *et al.*, 2001; Polwiang, 2020; Rezza *et al.*, 2007; N. Singh *et al.*, 2015). Several widely dispersed plant species (e.g., *Prosopis juliflora* (mesquite), *Parthenium hysterophorus* (parthenium weed)) significantly contribute to *Anopheles* mosquito longevity, and thereby enhance malaria transmission potential (Muller *et al.*, 2017; Nyasembe *et al.*, 2015; Tyagi *et al.*, 2015).

Some of the most widely dispersed invasive alien plants cause direct or indirect adverse effects. *Hedera helix* (ivy), native to Europe, is established in Australia, New Zealand, Hawaii, Brazil, and North America, where it causes allergic contact dermatitis (Bregnbak *et al.*, 2015; J. M. Jones *et al.*, 2009). The pollen of *Ambrosia artemisiifolia* (common ragweed), a native plant to Central and Northern America that has spread widely, is a common seasonal source of aeroallergens, and a major concern for public health, causing allergic rhinitis, fever, or dermatitis (Déchamp, 1999; Möller *et al.*, 2002). A single *Ambrosia artemisiifolia* plant can indeed release up to one billion pollen grains per season, and as low as 10 pollen grains per cubic meter of air can trigger an allergic reaction (DellaValle *et al.*, 2012; Emberlin, 1994; Fumanal *et al.*, 2007). High pollen exposure or volume may lead to increases in sensitization rate (Gabrio *et al.*, 2010; Jäger, 2000). *Prosopis juliflora* (mesquite) pollen also elicits highly allergenic reactions (Al-Frayh *et al.*, 1999; Ezeamuzie *et al.*, 2000; Kathuria & Rai, 2021; Killian & McMichael, 2004). *Heracleum mantegazzianum* (giant hogweed), native to southern Russia and Georgia but now spread throughout northern Europe, poses threats to human health due to its photoallergic properties, resulting from the intensely toxic furanocoumarin in its sap **(Figure 4.42)**. Contact with the plant, followed by sun exposure, may lead to the development of blisters and symptoms of burns (Carlsen & Weismann, 2007; Jakubska-Busse *et al.*, 2013; Klimaszyk *et al.*, 2014; Lim *et al.*, 2021).

Health impacts caused by invasive alien marine species have been amply documented. In the Mediterranean Sea, for example, the venomous *Rhopilema nomadica* (nomad jellyfish; **Figure 4.42**), *Pterois miles* (lionfish), and the lethally poisonous *Lagocephalus sceleratus* (silver-cheeked toadfish)

Box 4 16 Invasive alien species as disease vectors or reservoir hosts.

Beyond the health impacts discussed in **Box 4.15**, many invasive alien species can act as disease vectors (i.e., introducing parasites and pathogens to new regions along with their host, passing diseases directly to humans), reservoir hosts (where a disease can survive before a vector passes it onward), or facilitators (i.e., helping the occurrence of pathogen or vector).

Global trade in livestock, wildlife and plants is a key driver facilitating both intended and unintended introductions of pathogens, hosts, and vectors to new land areas, increasing the rate of disease emergence and health impacts on human populations (Bezerra-Santos *et al.*, 2021; Chinchio *et al.*, 2020; Fèvre *et al.*, 2006; Lounibos, 2002; Vilà *et al.*, 2021).

Diseases such as the bubonic plague, caused by the flea- and rat-borne bacterium *Yersinia pestis* (black death), have caused traumatic social and political upheavals (Athni *et al.*, 2021; Kosoy & Bai, 2019; Wells *et al.*, 2015). Mosquito species such as *Aedes aegypti* (yellow fever mosquito) and *Aedes albopictus* (Asian tiger mosquito) have spread since the fifteenth century, largely due to shipping, air and road transport and trade (Lounibos, 2002). These species have exacerbated the spread of the lethal yellow fever, dengue fever, chikungunya and Zika viruses, and other infectious diseases, throughout the Americas, Asia and, more recently Europe (Juliano & Lounibos, 2005; LaPointe, 2021; Romi *et al.*, 2018). *Culex quinquefasciatus* (southern house mosquito), a vector for lymphatic filariasis, St. Louis Encephalitis virus, and West Nile virus, has spread from West Africa, killing over a million people a year (LaPointe, 2021; Lounibos, 2002; Romi *et al.*, 2018). Invasive mammals and birds can alter the epidemiology of resident pathogens and become reservoir hosts, increasing disease risk for humans (Capizzi *et al.*, 2018). Most zoonotic human diseases are known to originate from mammals: rodents and bats are vectors for a high number of pathogens (Han *et al.*, 2016), and so are *Nyctereutes procyonoides* (raccoon dog), implicated in rabies and tapeworm transmission, and *Procyon lotor* (raccoon), implicated in roundworm transmission (Lojkić *et al.*, 2021; Page *et al.*, 2016). Introduced bird species, in particular psittaciform (parrots), columbiform (pigeons) and anseriform (duck) species, represent a hazard to good quality of life. Main zoonoses include psittacosis, cryptococcosis, listeriosis and salmonellosis, transmitted by direct contact or *via* insect vectors (fleas, lice, ticks and mites). Some galliform species, introduced for hunting, can cause salmonellosis and other gastroenteric diseases (Mori *et al.*, 2018).

The magnitude of risks and impacts arising from co-invasive pathogens is difficult to discern because few data exist on the links among invasive alien species, their parasites or pathogen load and zoonotic diseases (Hulme, 2014). Robust documentation of the prevalence and abundance of parasites, pathogens, and vectors of human diseases associated with high-risk alien hosts would be needed to initiate effective management.

Figure 4 42 Examples of invasive alien species causing serious health problems.

Lissachatina fulica (giant African land snail, top left), *Solenopsis invicta* (red imported fire ant, top right), *Heracleum mantegazzianum* (giant hogweed, bottom left), *Rhopilema nomadica* (nomad jellyfish, bottom right). Photo credits: Mark Brandon, Shutterstock – Copyright (top left) / Alexander Wild, WM Commons – Public domain (top right) / MurielBendel, WM Commons – CC BY-SA 4.0 (bottom left) / Jimmy, Adobe Stock – Copyright (bottom right).

present well-known dangers (Galil, 2018). With rising seawater temperature, it is likely these thermophilic species will expand their range. Though published records attest to the increasing spread and abundance of these species in the Mediterranean Sea, only fragmentary information is available concerning the spatial and temporal trends (**Glossary**) of their impacts (Bédry *et al.*, 2021; Galil, 2018). Even for common, wide-spread species with acute symptoms such as *Lagocephalus sceleratus* and *Rhopilema nomadica*, incidents are poorly documented. Öztürk and İşinibilir (2010) reported that in summer 2009, nomad jellyfish envenomation caused 815 hospitalizations in Turkey, but no data is available for other years and other locations. A similar pattern emerges from the records of toadfish poisoning, reported mainly in local journals and digital media (Ben Souissi *et al.*, 2014). The lack of region-wide, quantitative data on medically-treated health impacts could lead on one hand to medical errors (Beköz *et al.*, 2013), and on the other, prejudice risk analyses undertaken by management. Incidents involving large numbers of patients may be expected to become more frequent with changing environmental conditions, unless this becomes a public health priority (Glatstein *et al.*, 2018).

4.5.1.4 Impacts on human health: links with impacts on other constituents of good quality of life

Constituents of good quality of life are often linked, and impacts on human health are one type of impacts on other constituents of good quality of life. Many Indigenous Peoples and local communities experience these connections acutely due to their close physical and spiritual interactions with the environment **(Box 4.17)**. Many invasive alien species impact Indigenous Peoples and local communities' lifestyles, by restricting access to lands and participation in traditional activities (IPBES, 2020). For example, in Australia, *Anoplolepis gracilipes* (yellow crazy ant) and *Solenopsis invicta* (red imported fire ant), which can bite people, have prevented some Indigenous Peoples and local communities from taking part in traditional activities. *Pastinaca sativa* (parsnip), an invasive alien plant in Canada, has been documented by Indigenous Peoples and local communities to cause skin to become sensitive to sunlight, burning the skin, which is a problem for hunters of that community (IPBES, 2020). The increase of Lyme disease-bearing *Ixodes scapularis* (blacklegged or deer ticks) populations in Canada is indirectly impacting knowledge transmission as Indigenous Peoples and local communities are concerned about taking children out on the land, where the majority of Indigenous and local knowledge learning takes place (IPBES, 2020). Indigenous Peoples and local communities in Siberia have known that *Heracleum pubescens* (Sosnowskyi's hogweed or Borshchievik in Russian) has been a problem for them since the 1980s (IPBES, 2020); they report that it is highly poisonous when over 60cm high and seeding, with stems and leaves causing allergic reactions, severe dermatitis and may cause cancerous tumours, congenital malformations and even fatalities in humans and animals (IPBES, 2020). Indigenous Peoples and local communities often try to maintain access to land and carry out traditional activities, particularly passing knowledge onto the next generation. When invasive alien species impact upon these activities, this can lead to "cultural erosion" (Pfeiffer & Voeks, 2008), whereby knowledge, particularly names of native species, their habitats, and their cultural values and stories are not passed down to younger people, which can have negative implications for the good quality of life of future generations (Robin *et al.*, 2022; **Box 4.17**).

Box 4 17 The impacts on cultural species, cultural sites, cultural relationships and health of Indgenous Peoples and local communities, revealed through Indigenous and local knowledge and cross-cultural research in Australia's Northern Territory.

Cross-cultural research (using methodologies from different knowledge systems), was used in Arnhem Land, at the northeast corner of Australia's Northern Territory, to investigate invasive ungulates (buffalo, donkeys, pigs, cattle and horses) trampling and grazing on traditional bush food resources and impacting water quality at several culturally significant wetlands.

Wetlands provide Indigenous Peoples with drinking water, medicines and bush foods, including *Eleocharis dulcis* (Chinese water chestnut) and *Nymphaea* spp. (water lilies), and is host to aquatic fauna, including *Chelodina rugosa* (northern snake-necked turtle), which is an important seasonal source of protein (Fordham *et al.*, 2006; Ens, Fisher, *et al.*, 2015). The Indigenous People of Ngukurr, Arnhem Land, have raised concerns about drinking water from wetlands due to potential microbial contamination from feral invasive ungulates, yet deoxyribonucleic acid (DNA) analysis for the waterborne pathogens *Cryptosporidium* and *Giardia* revealed the latter was only detected in the late dry season and former was not detected at all (S. Russell *et al.*, 2021). The presence of invasive ungulates negatively impacts on indigenous access to bush food, medicine and freshwater resources which then reduces opportunities for cultural and spiritual practices (S. Russell *et al.*, 2020, 2021). For example, hooved ungulates damage *Chelodina rugosa* aestivating over the dry season (S. Russell *et al.*, 2021), and feral pig predation depletes turtle stocks immediately before Aboriginal harvesting (Fordham *et al.*, 2006). An Indigenous knowledge holder described an eco-cultural regime shift of a wetland ecosystem from a water lily (*Nymphaea violacea* and *Nymphaea macrosperma*) dominated

Box 4 17

system to a turbid, sediment dominated system. This was attributed to human depopulation of traditional lands and waters and the subsequent invasion by feral ungulates. Evidence for this "regime shift" is based on Indigenous ecological knowledge (S. Russell *et al.*, 2021). Transformation of this ecosystem has had implications for access to bush food resources; *Nymphaea* spp. Roots, stems, and bulbs that were a staple food for local Indigenous Peoples.

Although invasive ungulates are impacting ecological condition, indigenous cultural practice, and potentially human health, these animals present a significant food source and potential source of income to remote living Indigenous Peoples and local communities who have low socio-economic status (C. J. Robinson *et al.*, 2005). The conflicting impacts and benefits of these invasive alien species has meant that widespread and sustained control has not occurred across northern Australia. To accommodate the multiple values of these invasive alien species, at present, members of the Ngukurr community prefer maintenance of multi-functional landscapes where the multiple values of these species can be supported (Ens, Fisher, *et al.*, 2015). However, with economic development of this region, support of invasive ungulate management may increase **(Figure 4.43)**.

Figure 4 43 **Invasive ungulates pollute wetlands in Australia's Northern Territory; and indigenous rangers document the impacts on water quality and cultural species, including *Nymphaea* spp. (water lilies).**

Photo credit: Shaina Russell – CC BY 4.0.

4.5.2 Documented impacts of invasive alien species on good quality of life by realm

Impacts of invasive alien species on good quality of life vary by realm and unit of analysis. As a general pattern, the impact database developed through this chapter reveals that impacts are most often experienced through changes to material and immaterial assets, followed by impacts on health and social and cultural relationships.[14] Impacts on safety and freedom of choice and action are the two least documented components of good quality of life.

Notably, a small number of the units of analysis account for most of the impacts on good quality of life (**sections 4.5.2.1** and **4.5.2.2**). Among the documented negative impacts, 74 per cent occur in cultivated areas (including cropping, intensive livestock farming, etc.; 935 negative impacts caused by 332 invasive alien species), urban/ semi-urban (606 negative impacts caused by 245 invasive alien species), inland surface water and water bodies/ freshwater (411 negative impacts caused by 151 invasive alien species), and tropical and subtropical dry and humid forests (415 negative impacts caused by 136 invasive alien species). Similarly, 74 per cent of all positive impacts occur in cultivated areas (151 positive impacts caused by 79 invasive alien species), tropical and subtropical dry and humid forests (118 positive impacts caused by 70 invasive alien species), temperate and boreal forests and woodlands (82 positive impacts caused by 40 invasive alien species), and inland surface water and water bodies/freshwater (80 positive impacts caused by 49 invasive alien species). Examining positive impacts helps put these percentages

14. Data management report available at: https://doi.org/10.5281/zenodo.5766069

into perspective. The three least affected units of analysis with positive impacts, tundra and high mountain habitats (5 positive impacts, caused by 2 invasive alien species), Mediterranean forests, woodlands and scrub (2 positive impacts caused by 2 invasive alien species), and open ocean pelagic systems (1 positive impact caused by 1 invasive alien species), make up less than 2 per cent of all positive documented impacts. **Table 4.24** presents the main invasive alien species causing impacts on good quality of life in each unit of analysis.

Table 4 24 **Invasive alien species most frequently documented to cause negative or positive impacts on good quality of life by unit of analysis.**

A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Plants: Invertebrate: Vertebrate: Microorganisms:

Unit of analysis		List of invasive alien species with negative impacts on good quality of life (# of observations)		List of invasive alien species with positive impacts on good quality of life (# of observations)
Tropical and subtropical dry and humid forests	Microorganism	Dengue virus (76)	Invertebrate	*Subulina octona* (thumbnail awlsnail) (9)
	Invertebrate	*Lissachatina fulica* (giant African land snail) (36)	Invertebrate	*Gulella bicolor* (two-tone gulella) (8)
	Invertebrate	*Laevicaulis alte* (tropical leatherleaf slug) (11)	Invertebrate	*Allopeas clavulinum* (Spike awlsnail) (7)
			Invertebrate	*Allopeas gracile* (Graceful awlsnail) (7)
			Invertebrate	*Bradybaena similaris* (Asian trampsnail) (7)
			Invertebrate	*Deroceras leave* (meadow slug) (7)
			Invertebrate	*Gastrocopta servilis* (wandering snag) (7)
			Invertebrate	*Geostilbia aperta* (obtuse awlsnail) (7)
			Invertebrate	*Guppya gundlachi* (glossy granule) (7)
Temperate and boreal forests and woodlands	Invertebrate	*Agrilus planipennis* (emerald ash borer) (35)	Vertebrate	*Equus ferus* (wild horse) (4)
	Microorganism	*Phytophthora ramorum* (sudden oak death) (34)	Vertebrate	*Cervus nippon* (sika) (2)
	Microorganism	*Hymenoscyphus fraxineus* (ash dieback) (25)	Invertebrate	*Corythucha arcuata* (oak lace bug) (3)
Mediterranean forests, woodlands and scrub	Microorganism	*Xylella fastidiosa* (Pierce's disease of grapevines) (15)	Plant	*Agave americana* (century plant) (1)
	Microorganism	*Ceratocystis platani* (canker stain of plane) (13)		
	Microorganism	*Seiridium cardinale* (cypress canker) (11)		
Tundra and High Mountain habitats	Plant	*Conium maculatum* (poison hemlock) (1)	Vertebrate	*Equus ferus* (wild horse) (4)
	Plant	*Pinus* spp. (pine) (1)	Plant	*Melilotus albus* (honey clover) (1)
	Vertebrate	*Equus ferus* (wild horse) (1)		
Tropical and subtropical savannas and grasslands	Vertebrate	*Corvus splendens* (house crow) (9)	Plant	*Acacia mearnsii* (black wattle) (1)
	Plant	*Acacia mangium* (brown salwood) (3)	Plant	*Centaurea solstitialis* (yellow starthistle) (1)
	Plant	*Cenchrus biflorus* (Indian sandbur) (3)	Plant	*Hyparrhenia rufa* (jaragua grass) (1)
			Plant	*Prosopis juliflora* (mesquite) (1)
			Plant	*Tithonia* spp. (1)

Table 4 24

Unit of analysis	List of invasive alien species with negative impacts on good quality of life (# of observations)	List of invasive alien species with positive impacts on good quality of life (# of observations)
Temperate Grasslands	*Lonchura oryzivora* (Java sparrow) (10)	*Sporobolus anglicus* (common cordgrass) (1)
	Acridotheres tristis (common myna) (8)	*Rosa rugosa* (rugosa rose) (8)
	Corvus splendens (house crow) (8)	*Bombus terrestris* (bumble bee) (1)
		Columba livia (pigeons) (10)
		Alces alces (moose) (1)
		Bos taurus (cattle) (1)
		Capra hircus (goats) (1)
		Cervus elaphus canadensis (elk) (1)
		Cervus elaphus (red deer) (1)
Deserts and xeric shrublands	*Cenchrus ciliaris* (buffel grass) (7)	*Prosopis juliflora* (mesquite) (9)
	Prosopis spp. (5)	*Opuntia* spp. (pricklypear) (2)
	Camelus spp. (camels) (4)	*Prosopis glandulosa* (honey mesquite) (2)
Urban/Semi-urban	*Solenopsis invicta* (red imported fire ant) (38)	*Columba livia* (pigeons) (10)
	Lissachatina fulica (giant African land snail) (32)	*Corvus splendens* (house crow) (2)
	Monomorium pharaonis (pharaoh ant) (16)	*Sarasinula plebeia* (Caribbean leatherleaf slug) (7)
		Trichocorixa verticalis (water boatman) (2)
Cultivated areas (incl. cropping, intensive livestock farming etc.)	*Spodoptera frugiperda* (fall armyworm) (46)	*Columba livia* (pigeons) (10)
	Bactrocera dorsalis (Oriental fruit fly) (40)	*Subulina octona* (thumbnail awlsnail) (9)
	Phenacoccus manihoti (cassava mealybug) (35)	*Gulella bicolor* (two-tone gulella) (8)
Aquaculture areas	*Cyprinus carpio* (common carp) (7)	*Azolla filiculoides* (water fern) (1)
	Oreochromis niloticus (Nile tilapia) (6)	*Oreochromis mossambicus* (Mozambique tilapia) (3)
	Hypophthalmichthys molitrix (silver carp) (4)	*Clarias gariepinus* (North African catfish) (1)
	Oreochromis mossambicus (Mozambique tilapia) (4)	*Oreochromis niloticus* (Nile tilapia) (1)
		Poecilia reticulata (guppy) (1)
Wetlands – peatlands, mires, bogs	*Elaeagnus angustifolia* (Russian olive) (3)	*Trichocorixa verticalis* (water boatman) (2)
	Acridotheres tristis (common myna) (2)	*Acridotheres javanicus* (Javan myna) (2)
	Threskiornis aethiopicus (sacred ibis) (2)	*Threskiornis aethiopicus* (sacred ibis) (2)
Inland surface waters and water bodies/freshwater	*Dreissena polymorpha* (zebra mussel) (19)	*Oreochromis niloticus* (Nile tilapia) (6)
	Cyprinus carpio (common carp) (19)	*Oreochromis mossambicus* (Mozambique tilapia) (5)
	Oreochromis niloticus (Nile tilapia) (14)	*Lates niloticus* (Nile perch) (4)
		Procambarus clarkii (red swamp crayfish) (4)

Table 4.24

Unit of analysis	List of invasive alien species with negative impacts on good quality of life (# of observations)	List of invasive alien species with positive impacts on good quality of life (# of observations)
Shelf ecosystems (neritic and intertidal/littoral zone)	*Rhopilema nomadica* (nomad jellyfish) (10)	*Eucheuma denticulatum* (eucheuma seaweed) (7)
	Gonionemus spp. (4)	*Kappaphycus alvarezii* (elkhorn sea moss) (3)
	Lagocephalus sceleratus (silver-cheeked toadfish) (5)	*Paralithodes camtschaticus* (red king crab) (2)
	Plotosus lineatus (striped eel catfish) (4)	
Open ocean pelagic systems (euphotic zone)	*Pterois* spp. (3)	*Pterois* spp. (1)
Coastal areas intensively used for multiple purposes by humans	*Dreissena polymorpha* (zebra mussel) (10)	*Corbicula fluminea* (Asian clam) (2)
	Asterias amurensis (northern Pacific seastar) (3)	*Petromyzon marinus* (sea lamprey) (2)
	Dreissena rostriformis bugensis (quagga mussel) (3)	*Salmo trutta* (brown trout) (2)
	Petromyzon marinus (sea lamprey) (3)	

4.5.2.1 Patterns of negative and positive impacts of invasive alien species on good quality of life in the terrestrial realm

The terrestrial realm accounts for most of the documented impacts on good quality of life compared to aquatic realms; with 82 per cent (2,629 impacts) of all negative impacts and 81 per cent (467 impacts) of positive impacts. This is consistent with the literature on impacts of invasive alien species on livelihood, which indicates that a greater proportion of studies focused on terrestrial ecosystems have been conducted in savanna and woodland environments compared to freshwater ecosystems (R. T. Shackleton, Shackleton, *et al.*, 2019).

Most impacted units of analysis in the terrestrial realm

Cultivated areas have the highest number of documented invasive alien species (332) and negative impacts (935) **(Table 4.25)**. This pattern may be attributed in part to the greater attention paid to cultivated areas in research, given the critical role of food security in meeting basic human needs. Additionally, the ease of measuring human access to food in cultivated areas may make them a more accessible unit of analysis than other ecosystems (P. L. Howard, 2019; Pimentel *et al.*, 2005). Large numbers of invasive alien species have also been documented in urban/semi-urban areas (245 invasive alien species), tropical and subtropical dry and humid forests (136 invasive alien species), and temperate and boreal forests and woodlands (101 invasive alien species), which, together, round out the top four affected units of analysis, accounting for 83 per cent of all documented negative impacts in the terrestrial realm. Tundra and high mountain habitats and deserts and xeric shrublands are the least affected units of analysis, and only host 5 per cent of the documented invasive alien species causing negative impacts on good quality of life in the terrestrial realm.

Material and immaterial assets are the most impacted constituent of good quality of life across units of analysis. However, in deserts and xeric shrublands, health is the dominant constituent negatively impacted by invasive alien species. There are very few documented impacts on safety across most units of analysis.

Invasive alien taxa most often documented causing negative impacts on good quality of life in the terrestrial realm

The most prominent taxa negatively impacting good quality of life differ across units of analysis **(Table 4.24)**. For example, cultivated areas, temperate grasslands, and urban/semi-urban areas are mainly affected by invasive alien animals such as *Spodoptera frugiperda* (fall armyworm) that causes significant damage to agriculture and rice crops (Kumela *et al.*, 2019; **Box 4.18**). These species can create additional pressures for farmers in Africa or native American lands where the agricultural sector struggles to support farmers' livelihoods because of the lack of ownership, low or no financial capital, and increasing risks due to climate change (Gautam *et al.*, 2013; P. L. Howard, 2019). Temperate grasslands contended with *Acridotheres tristis* (common myna) whose droppings can irritate

Table 4 25 **Negative impacts on good quality of life in the terrestrial realm.**

Darker colours indicate higher documented numbers of invasive alien species or impacts. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Unit of analysis	Material and immaterial assets		Freedom of choice and action		Health		Social and cultural relationships		Safety	
	species	impacts	species	impacts	species	impacts	species	impacts	species	impacts
Tropical and subtropical dry and humid forests	66	213	7	14	41	143	17	37	5	8
Temperate and boreal forests and woodlands	48	183	7	10	24	54	15	34	7	15
Mediterranean forests, woodlands and scrub	11	77	0	0	3	3	0	0	1	1
Tundra and High Mountain habitats	1	1	0	0	1	1	1	1	0	0
Tropical and subtropical savannas and grasslands	18	33	4	4	20	28	13	21	2	2
Temperate Grasslands	54	103	0	0	12	32	3	4	0	0
Deserts and xeric shrublands	5	11	0	0	6	11	4	5	5	8
Wetlands – peatlands, mires, bogs	5	8	4	7	5	9	3	7	0	0
Urban/Semi-urban	111	299	2	3	89	240	31	48	12	16
Cultivated areas (incl. cropping, intensive livestock farming etc.)	224	730	21	26	56	129	22	38	9	12

people's skin and lungs (Peacock *et al.*, 2007). Urban and semi-urban areas are most impacted by the aggressive *Solenopsis invicta* (red imported fire ant), whose powerful sting can cause injury and death to people, wildlife, and pets (Gutrich *et al.*, 2007). Their ability to tunnel also impacts infrastructure, such as roads, power distribution systems, and irrigation systems (Gutrich *et al.*, 2007). Impacts on good quality of life in Mediterranean forests, woodlands and scrub are mostly caused by microbes, such as *Xylella fastidiosa* (Pierce's disease of grapevines), known for killing olive trees (Schneider *et al.*, 2020), reiterating that material and immaterial assets are the most affected component of good quality of life.

The remaining units of analysis experience impacts from several taxa. Tropical and subtropical dry and humid forests and temperate and boreal forests and woodlands are negatively affected by microbes and animals. Deserts and xeric shrublands, tropical and subtropical savannas and grasslands, tundra and high mountain, and wetland habitats contend with both plants and animals. In the deserts and xeric shrublands in Africa, *Prosopis* spp. (mesquite) threatens safety through impacts on personal safety and secure resource access as the invasion has forced large predators like lions to move closer to villages, leading to livestock and human deaths (P. L. Howard, 2019). There are no units of analysis exclusively impacted by plant species.

Box 4 18 ***Spodoptera frugiperda*** **(fall armyworm) – how impacts on nature's contributions to people and impacts of management affect good quality of life for Indigenous Peoples and local communities.**

Impacts on nature's contributions to people: crop losses

A majority (61 per cent) of the studies reviewed[15] have documented that Indigenous Peoples and local communities suffer yield losses due to the invasion of *Spodoptera frugiperda* (fall armyworm). The crop yield loss estimates due to *Spodoptera frugiperda* range from 10 per cent in Malawi (Murray *et al.*, 2019) to as high as 58 per cent in Zimbabwe (Chimweta *et al.*, 2020; **Table 4.26**). Most of the yield loss estimates are related to maize production, but the FAO also

15. Data management report available at: https://doi.org/10.5281/zenodo.5760266

Box 4 18

found that *Spodoptera frugiperda* has caused 6 per cent and 2 per cent millet and sorghum production losses, respectively, at the national level in Namibia (FAO, 2018). There is also evidence suggesting that the yield loss estimates were higher in the early years of the *Spodoptera frugiperda* invasion. For instance, Day *et al.* (2017) found maize yield losses of 45 per cent and 40 per cent in Ghana and Zambia respectively (or 8.3 to 20.6 million tonnes annually in 12 African countries), but a follow-up study a year later by Rwomushana *et al.* (2018) showed maize yield losses of 26 per cent and 35 per cent in the two respective countries (or 4.1 to 17.7 million tonnes annually in 12 African countries). As noted by Rwomushana *et al.* (2018), this decline in yield losses could be due to build-up of natural enemies, climatic factors, improved management or the possibility that farmers are getting better at estimating *Spodoptera frugiperda*-induced yield loss. It should be mentioned that most of the yield loss estimates were based on farmers' perceptions, which may have overestimated true losses (Baudron *et al.*, 2019) even when controlling for potential confounding factors in a regression framework, documented *Spodoptera frugiperda*-induced yield losses are nearly 12 per cent (Baudron *et al.*, 2019; Kassie *et al.*, 2020).

Table 4 26 **Yield loss estimates due to *Spodoptera frugiperda* (fall armyworm) invasion.**

A data management report for the literature review underpinning this table is available at https://doi.org/10.5281/zenodo.5760266

Study	Country	Yield loss estimates
Asare-Nuamah (2022)	Ghana	Massive (no exact estimate)
Bariw *et al.* (2020)	Ghana	17.2 per cent
Baudron *et al.* (2019)	Zimbabwe	11.6 per cent
Chimweta *et al.* (2020)	Zimbabwe	58 per cent
Day *et al.* (2017)	Ghana and Zimbabwe	45 per cent in Ghana; 40 per cent in Zambia (extrapolated to up to 20.6 million tonnes annually in 12 Africa countries)
De Groote *et al.* (2020)	Kenya	33 per cent or 1 million tonnes
FAO (2018)	Namibia	14 per cent of maize (8 per cent in communal areas and 6 per cent in commercial farms); 6 per cent of millet; 2 per cent of sorghum
Turot *et al.* (2019)	Tanzania	10.8 per cent at area level; 15.8 per cent at farm level
Girsang *et al.* (2020)	Indonesia	26.6 per cent
Houngbo *et al.* (2020)	Benin	49 per cent
Kansiime *et al.* (2019)	Zambia	28 per cent
Kassie *et al.* (2020)	Ethiopia	11.5 per cent
Koffi *et al.* (2020)	Ghana	132,450 tons in 2016; 180,000 tons in 2017; 36,000 tons in 2018
Kumela *et al.* (2019)	Ethiopia and Kenya	46.5 per cent in Ethiopia; 38.8 per cent in Kenya
Mayee *et al.* (2021)	India	Decline in maize area from 9.2 million ha in 2018 to 8.2 million ha in 2019 (no exact estimate)
Murray *et al.* (2021)	Kenya	Up to 50 per cent
Murray *et al.* (2019)	Malawi	10 per cent
Nyangau *et al.* (2020)	Kenya and Uganda	No exact estimate
Rwomushana *et al.* (2018)	Ghana and Zambia	26 per cent in Ghana; 35 per cent in Zambia) extrapolated to up to 17.7 million tonnes annually in 12 African countries)
van Loon *et al.* (2019)	Ghana	Severe (no exact estimate)

Box 4 18

Impacts on good quality of life

Spodoptera frugiperda caused crop yield loss in invaded systems and has consequently resulted in increased production costs, decline in farmers' income, hunger and worsened food insecurity **(Figure 4.44)**. For example, Girsang *et al.* (2020) found that *Spodoptera frugiperda* led to 50 per cent and 71.4 per cent per cent increase in labour and pesticide costs in 2019, respectively, in North Sumatra province of Indonesia. Similarly, it is estimated that farmer's expenditure on pesticides has increased by US$195 per hectare (241 per cent) due to the *Spodoptera frugiperda* invasion in China's Yunnan province (Yang *et al.*, 2021). Moreover, Kassie *et al.* (2020) found that *Spodoptera frugiperda* invasion was associated with a 25 per cent reduction in maize sales in southern Ethiopia, while Tambo *et al.* (2021) documented a reduction in per capita household income by 44 per cent and a 17 per cent higher likelihood of hunger in Zimbabwe due to severe levels of *Spodoptera frugiperda* infestation.

The *Spodoptera frugiperda* outbreak is also having negative impacts on the livestock sector in terms of reduced availability of livestock feed, such as stover, grains, straw and pasture land (FAO, 2018; Mayee *et al.*, 2021). The Indian government imported 130,000 tonnes of maize in 2019 for the poultry industry as a result of a reduction in maize production (Mayee *et al.*, 2021).

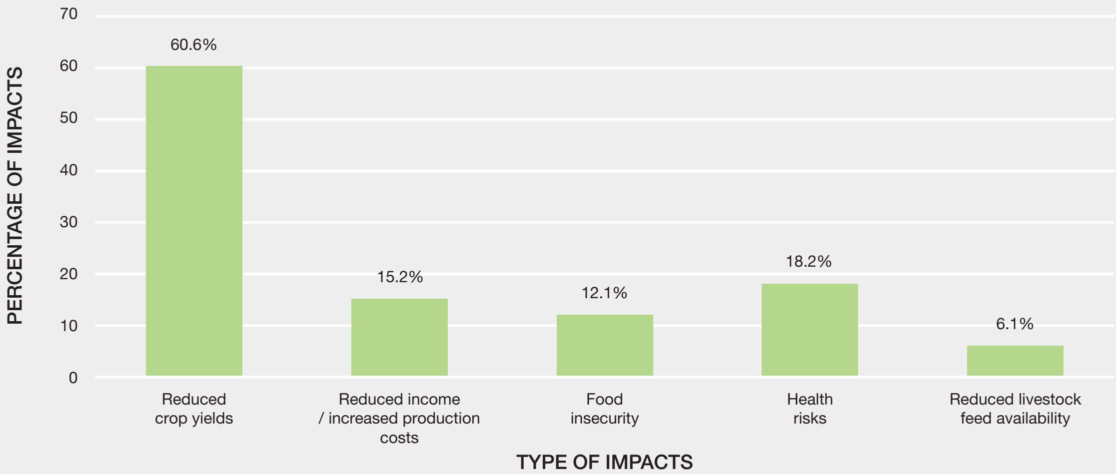


Figure 4 44 **Percentage of documented impacts (y axis) of *Spodoptera frugiperda* (fall armyworm) on good quality of life of Indigenous Peoples and local communities.**

Spodoptera frugiperda negatively impacts crop yields, income and production costs, food security, health and livestock feed availability (x axis). The results are presented in percentages of the 33 case studies reviewed. A data management report for the literature review underpinning this figure is available at https://doi.org/10.5281/zenodo.5760266

The synthetic pesticides used by smallholders for *Spodoptera frugiperda* control have been shown to pose high risks to human health (Murray *et al.*, 2019, 2021; Kumela *et al.*, 2019). Several studies have shown that farmers who used pesticide to control *Spodoptera frugiperda* experience pesticide-related illness, such as dizziness, headache, skin and eye irritation and stomach ache (Kansiime *et al.*, 2019; Rwomushana *et al.*, 2018; Tambo *et al.*, 2020). The use of pesticides also has been shown to affect native species (Kumela *et al.*, 2019).

Positive impacts caused by invasive alien species on good quality of life in the terrestrial realm

As with negative impacts, positive impacts in the terrestrial realm are clustered within a few units of analysis **(Table 4.27)**. Cultivated areas are most positively impacted, with 79 invasive alien species causing 151 documented impacts. Tropical and subtropical dry and humid forests (70 invasive alien species causing 118 impacts) and temperate and boreal forests and woodlands (40 species causing 82 impacts) round out the top three positively impacted units of analysis. Taken together, these three units of analysis account for 75 per cent of documented terrestrial positive documented impacts. Meanwhile, the three least affected terrestrial units of analysis, Mediterranean forests, woodlands and scrub, tundra and high mountain habitats, and wetlands (peatlands, mires, bogs), account for only three per cent of positive terrestrial documented impacts.

Material and immaterial assets account for the most positively documented component of good quality of life across most units of analysis, followed by health, showing a similar pattern than for negative impacts. However, the order differs for tundra and high mountain habitats, tropical and subtropical savannas and grasslands, and temperate grasslands that are mainly impacted through positive changes to social and cultural relationships, followed by positive changes to material and immaterial assets. Safety is the least documented positively impacted component of good quality of life, accounting for only three per cent of positive terrestrial impacts.

Examining the positive impacts highlights the different ways invasive alien species interact with people across landscapes. Plants are one of the most documented taxa affecting Mediterranean forests, woodlands and scrubs (e.g., *Agave americana* (century plant)), deserts and xeric shrublands (e.g., *Prosopis juliflora* (mesquite)), and tropical and subtropical savannas and grasslands (e.g., *Acacia mearnsii* (black wattle)). This result mirrors findings from R. T. Shackleton, Shackleton, *et al.* (2019), who documented that most case studies on positive impacts on livelihoods involve invasive alien plants, often intentionally introduced. These plant species affect different components of good quality of life, which widely differ across units of analysis. For example, as a source of fuelwood, *Prosopis juliflora* is an important source of energy for cooking and heating, along with a possible source of income for those who sell the wood, which can cause significant positive impacts on assets. Human health is positively affected by species such as *Acacia mearnsii*, known for its antibacterial properties and effectiveness in treating illnesses as shigellosis (Olajuyigbe & Afolayan, 2012). Sociocultural relationships benefit from species such as *Agave americana* (century plant).

Tropical and subtropical dry and humid forests, temperate and boreal forests and woodlands, urban/semi-urban, cultivated areas, and wetlands (peatlands, mires, bogs), are mainly positively impacted by animals. Many of these species impact good quality of life either through material and immaterial assets, by providing a new way of creating or enhancing livelihood or through improvements to human health outcomes. *Equus ferus* (wild horse) in temperate and boreal forests and woodlands (and tundra and high mountain habitats) illustrates the prominent role some invasive alien species play in maintaining cultural identities, especially among Indigenous peoples and local communities (Bhattacharyya *et al.*, 2011; Bhattacharyya & Larson, 2014). The remaining units of analysis, tundra and high mountain habitats, temperate grasslands, benefit from a several invasive alien plants and animals. Importantly, there are no documented cases of microbes producing positive terrestrial impacts on good quality of life.

Table 4 27 **Positive impacts on good quality of life in the terrestrial realm.**

Darker colours indicate higher documented numbers of invasive alien species or impacts. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Invasive alien species	Material and immaterial assets		Freedom of choice and action		Health		Social and cultural relationships		Safety	
	species	impacts	species	impacts	species	impacts	species	impacts	species	impacts
Tropical and subtropical dry and humid forests	25	48	1	1	30	41	13	27	1	1
Temperate and boreal forests and woodlands	8	33	1	1	14	14	16	26	1	8
Mediterranean forests, woodlands and scrub	0	0	0	0	0	0	2	2	0	0
Tundra and High Mountain habitats	1	1	0	0	0	0	1	4	0	0
Tropical and subtropical savannas and grasslands	3	3	0	0	4	4	3	4	1	1
Temperate Grasslands	4	15	0	0	15	15	15	18	1	1
Deserts and xeric shrublands	3	9	1	1	1	1	3	3	1	2
Wetlands – peatlands, mires, bogs	1	1	0	0	3	5	2	2	0	0
Urban/Semi-urban	9	20	0	0	6	7	2	4	1	1
Cultivated areas (incl. cropping, intensive livestock farming etc.)	27	65	1	1	36	54	14	30	1	1

4.5.2.2 Patterns of negative and positive impacts of invasive alien species on good quality of life in the marine and inland waters realms

There are 575 documented negative impacts on good quality of life affecting the marine and inland waters realms.

Negative impact caused by invasive alien species on good quality of life across units of analysis in the marine and inland waters realms

Inland surface waters and water bodies are the most impaired of all aquatic units of analysis (151 invasive alien species causing 411 negative impacts), accounting for 71 per cent of all negative aquatic impacts **(Table 4.25)**. The least affected unit of analysis, open ocean pelagic systems, has only one documented invasive alien species, generating three negative impacts on material and immaterial assets, which accounts for less than one per cent of negative aquatic impacts.

The top two components of good quality of life most negatively affected across all aquatic domains are material and immaterial assets, followed by health **(Table 4.28)**. In shelf ecosystems (neritic, intertidal and littoral zone), health dominates followed by material and immaterial assets.

The top documented invasive alien species causing negative impacts on good quality of life in the aquatic realm are attributed solely to animals. Inland surface waters and water bodies are subject to impacts by invasive alien animals such as *Dreissena polymorpha* (zebra mussel) and *Cyprinus carpio* (common carp). For example, *Dreissena polymorpha*, which also impacts coastal areas intensively used for multiple purposes by humans, is known for its impacts on livelihoods and access to goods by clogging pipes used in water treatment plants, irrigation, and power generation stations (Elliott *et al.*, 2005). *Cyprinus carpio* limits access to nutritious food and adequate livelihoods by quickly dominating native fish species, negatively affecting fishing and recreation opportunities (Beardmore, 2015; A. K. Singh *et al.*, 2010). These two species highlight the numerous ways invasive alien species can negatively impact a single unit of analysis. Aquaculture areas are impacted by species such as *Oreochromis niloticus* (Nile tilapia) that negatively affect native fish and harms local fishermen's livelihoods (Ogutu-Ohwayo, 1990). Open ocean pelagic systems can be invaded by *Pterois* spp. (lionfish), which negatively affects commercially important native species (Johnston *et al.*, 2017). Health is the most impacted component of good quality of life in shelf ecosystems, with, for instance, *Rhopilema nomadica* (nomad jellyfish), known for its venomous stings (Öztürk & İşinibilir, 2010).

Positive impacts caused by invasive alien species on good quality of life across units of analysis in the marine and inland waters realm

The aquatic realm had 117 positive documented impacts caused by 73 invasive alien species. Inland surface waters and water bodies (49 invasive alien species, causing

Table 4 28 **Negative impacts on good quality of life in the marine and inland waters realms.**

Darker colours indicate higher documented numbers of invasive alien species or impacts. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Unit of analysis	Material and immaterial assets		Freedom of choice and action		Health		Social and cultural relationships		Safety	
	species	impacts	species	impacts	species	impacts	species	impacts	species	impacts
Aquaculture areas	38	69	0	0	4	4	0	0	1	1
Inland surface waters and water bodies/freshwater	64	224	16	41	32	83	23	44	16	19
Shelf ecosystems (neritic and intertidal/littoral zone)	11	15	0	0	14	35	0	0	0	0
Open ocean pelagic systems (euphotic zone)	1	3	0	0	0	0	0	0	0	0
Coastal areas intensively used for multiple purposes by humans	16	20	0	0	5	10	4	6	1	1

Table 4 29 **Positive impacts on good quality of life in the marine and inland waters realms.**

Darker colours indicate higher documented numbers of invasive alien species or impacts. A data management report for the database of impacts developed through this chapter is available at https://doi.org/10.5281/zenodo.5766069

Unit of analysis	Material and immaterial assets		Freedom of choice and action		Health		Social and cultural relationships		Safety	
	species	impacts	species	impacts	species	impacts	species	impacts	species	impacts
Aquaculture areas	5	7	0	0	0	0	0	0	0	0
Inland surface waters and water bodies/freshwater	32	59	1	1	6	10	7	7	3	3
Shelf ecosystems (neritic and intertidal/littoral zone)	5	13	0	0	1	1	0	0	0	0
Open ocean pelagic systems (euphotic zone)	1	1	0	0	0	0	0	0	0	0
Coastal areas intensively used for multiple purposes by humans	6	8	0	0	0	0	6	7	0	0

80 impacts) and Coastal areas intensively used for multiple purposes by humans (12 invasive alien species, causing 15 impacts) account for 81 per cent of aquatic realm positive impacts **(Table 4.29)**. Open ocean pelagic systems are the least impacted unit of analysis, with only one documented invasive alien species, *Pterois* spp. (lionfish). Material and immaterial assets, social/cultural relationships, and health are documented to be the most affected components of good quality of life. Impacts on safety are the least documented component, and only observed in inland surface waters and water bodies where, for instance, *Pontederia crassipes* (water hyacinth) assist with creating resilient communities by removing heavy and toxic metals from waterways (Dixit & Dhote, 2010).

While invasive alien plants are documented to cause more positive impacts than negative impacts (R. T. Shackleton, Shackleton, *et al.*, 2019), positive impacts are mostly caused by invasive alien animals in inland surface waters and water bodies/freshwater (e.g., *Oreochromis niloticus* (Nile tilapia)), in coastal areas intensively used for multiple purposes by people (e.g., *Corbicula fluminea* (Asian clam)), and in open ocean pelagic systems (e.g., *Pterois* spp. (lionfish)). Many of these positive impacts are due to changes in material and immaterial assets, such as creating new opportunities for income and recreation. Postive impacts on good quality of life in aquaculture and shelf ecosystems are mostly caused by invasive alien plants and animals. As with positive terrestrial impacts, there are no documented microbes causing positive impacts on good quality of life in the aquatic realm.

4.5.3 Documented impacts on good quality of life by region and taxonomic group

4.5.3.1 General patterns

Invasive alien species affect good quality of life in all regions. Several patterns of documented impacts emerge when examining the positive and negative impacts across regions. In particular, for both negative and positive impacts, the Asia-Pacific region has the most documented impacts, followed by Europe and Central Asia, the Americas, Africa, and Antarctica. The database of impacts developed through this chapter mirrors previous reports, showing that most impacts of invasive alien species on livelihoods are documented in the developing world, particularly southeast Asia (R. T. Shackleton, Shackleton, *et al.*, 2019). Negative impacts on good quality of life are mostly documented for material and immaterial assets and health. Safety is the least documented component of good quality of life. This result differs for positive impacts, where material and immaterial assets and social and cultural relationships are the most impacted components of good quality of life, while freedom of choice and action is least affected.

However, there are few consistent patterns when comparing the taxa that cause impacts across regions. For example, even though invertebrates cause the majority of negative impacts for most regions, the second, third, and fourth most prominent species vary for each region. The order of impacts by region is as follows, Africa: invertebrates, plants, vertebrates, microbes; Europe and Central Asia:

invertebrates, microbes, plants, vertebrates; Americas: invertebrates, vertebrates, plants, microbes; Asia-Pacific: invertebrates, plants, vertebrates, microbes; Antarctica: vertebrate only. In terms of positive impacts, plants generally have the most significant number of impacts documented for all regions, except for the Asia-Pacific region, where invertebrates have the highest number of invasive alien species with documented positive impacts. Finally, negative impacts tend to be more evenly distributed across regions when looking beyond the most impacted taxonomic group (i.e., the second, third or fourth most dominant taxonomic group). In contrast, positive impacts vary widely among taxa. This result follows (R. T. Shackleton, Shackleton, *et al.*, 2019), where the positive impacts of invasive alien species varied substantially between case studies and different species.

4.5.3.2 Patterns of negative impacts on good quality of life by taxonomic group and region

A total of 484 documented invasive alien species have caused negative impacts on good quality of life in Asia-Pacific, 347 in Europe and Central Asia, 296 in the Americas, 90 in Africa, and one in Antarctica **(Figure 4.45)**. Across almost all regions, change to material and immaterial assets is the most frequently documented negative impact on good quality of life. The highest number of documented negative impacts is found in Asia-Pacific (853 impacts), followed by Europe and Central Asia (598 impacts), Africa (286 impacts), and the Americas (265 impacts) **(Figure 4.46)**. Antarctica only has one documented impact on good quality of life, through health changes. Health impacts are the second most commonly documented impact on good quality of life for all other regions, with the highest number of impacts documented in Asia-Pacific (290 impacts). There are 223 documented negative impacts on health in the Americas, 139 in Europe and Central Asia, and 69 in Africa. Social and cultural relationships, such as environmental equity and social infrastructure, is the third most impacted component of good quality of life, which is relatively evenly distributed across Asia-Pacific (89 impacts), the Americas (81 impacts), and Europe and Central Asia (62 impacts). Impacts on safety, such as risks to personal safety and security from disasters, have been less documented, where the top two impacted regions are the Americas (42 impacts) and Asia-Pacific (27 impacts).

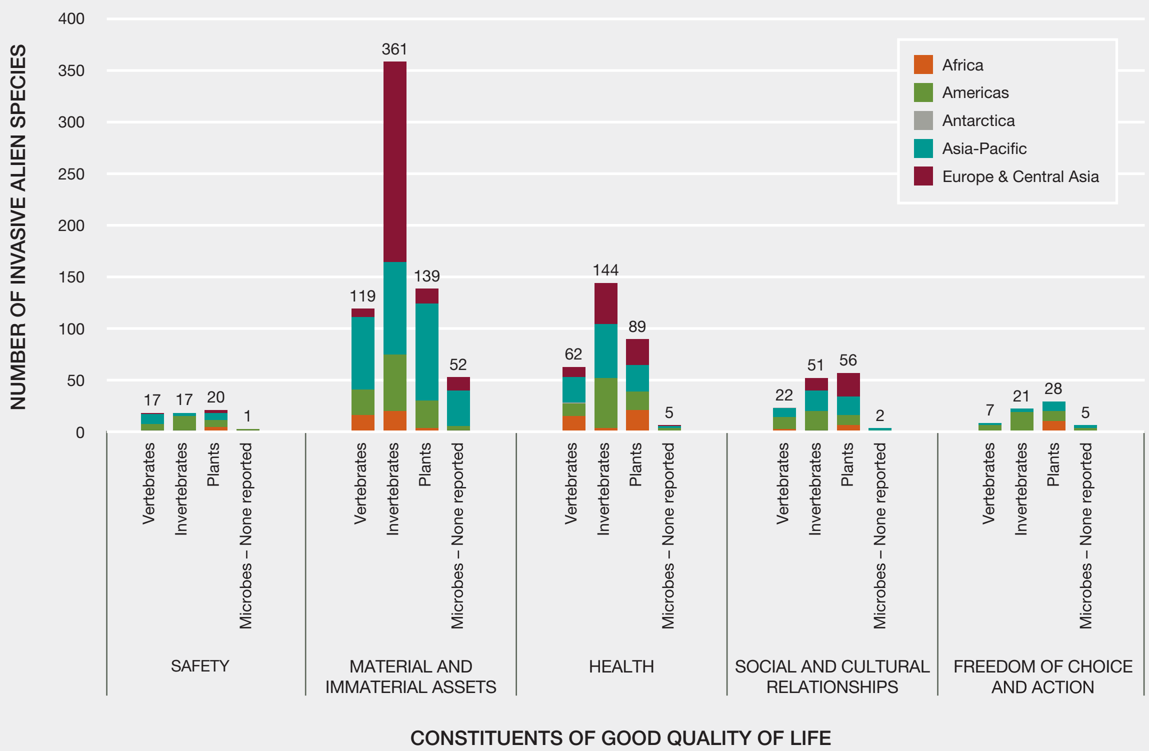


Figure 4 45 **Number of invasive alien species (y axis) causing negative impacts on constituents of good quality of life by taxonomic group and IPBES region (x axis).**

A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

The number of documented negative impacts on good quality of life for specific taxa varies by region, but some patterns do emerge (**Figures 4.45** and **4.46**). Invasive alien invertebrates are the main taxonomic group causing negative impacts on good quality of life across regions, with 594 species (51 per cent of all invasive alien species causing negative impacts on good quality of life). Negative impacts caused by invertebrates are relatively evenly distributed across regions: they have caused 494 negative impacts (31 per cent) in Europe and Central Asia, 457 in the Asia-Pacific region (30 per cent), 365 in the Americas (23 per cent), 258 in Africa (16 per cent), and none in Antarctica. Plants account for 21 per cent of all negative impacts across regions and are the second most documented taxonomic group affecting good quality of life in Asia-Pacific and Africa. More than half of negative impacts caused by plants on good quality of life are heavily concentrated in Asia-Pacific (391 impacts; 58 per cent of all impacts caused by plants on good quality of life). The remaining share of negative impacts are spread evenly across Europe and Central Asia (108 impacts; 16 per cent), the Americas (92 impacts; 14 per cent), and Africa (78 impacts; 12 per cent). There are no observed impacts of invasive alien plants in Antarctica **(Figure 4.46)**. Vertebrates cause 17 per cent of documented negative impacts on good quality of life across all regions, where documented impacts are heavily concentrated in the Asia-Pacific region (347 impacts; 64 per cent of all negative impacts caused by vertebrates). There are 116 impacts (21 per cent) caused by invasive alien vertebrates on good quality of life in the Americas, 49 in Africa (9 per cent), and 30 in Europe and Central Asia (6 per cent). Vertebrates are the least documented taxonomic group in Europe and Central Asia, and there is only one impact caused by a vertebrate documented in Antarctica **(Figure 4.46)**. Finally, microbes are the least documented taxonomic group to negatively impact good quality of life in most regions, accounting for 10 per cent of impacts across taxa **(Figure 4.46)**.

Europe and Central Asia record the majority of impacts caused by microbes across all regions (177 impacts; 52 per cent of all impacts caused by microbes). The Americas have the second-highest share of negative impacts caused by microbes (87 impacts; 26 per cent), closely followed by the Asia-Pacific region (74 impacts; 22 per cent). There are no documented microbes affecting good quality of life in Africa or Antarctica.

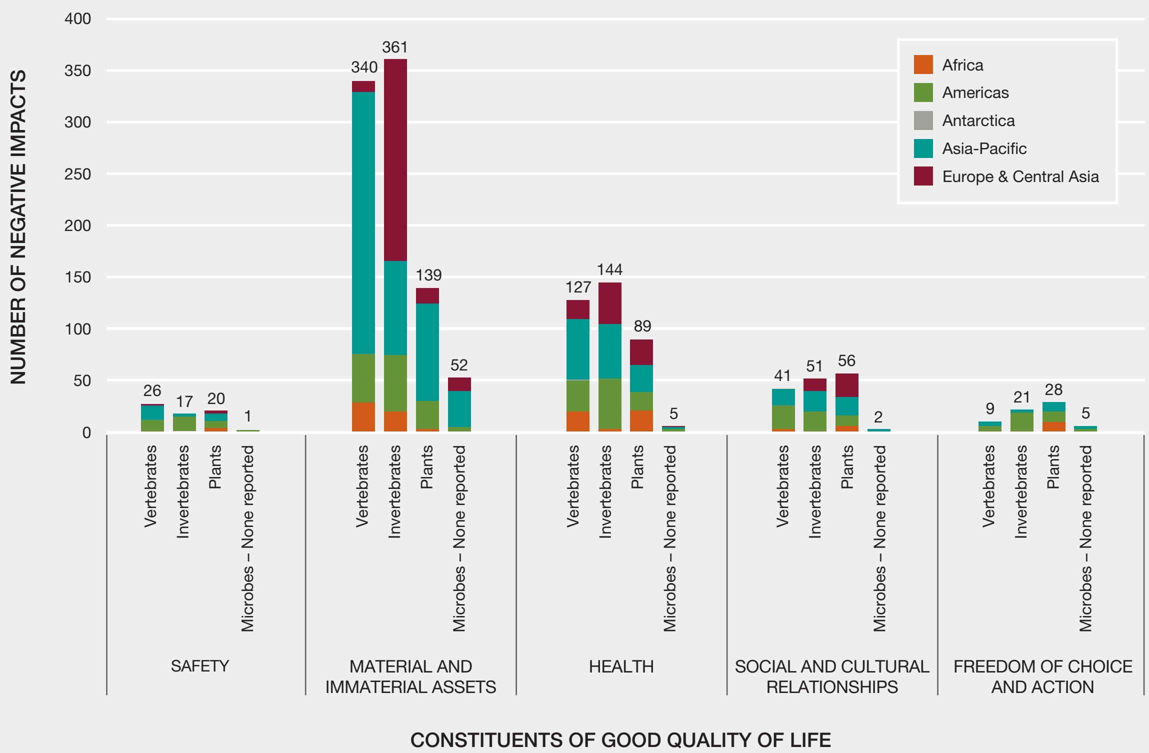


Figure 4 46 **Number of negative impacts (y axis) on constituents of good quality of life by taxonomic group and region (x axis).**

A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

4.5.3.3 Patterns of positive impacts on good quality of life by taxonomic group and region

There are 236 documented species that cause positive impacts on good quality of life, including 156 in the Asia-Pacific region, 56 in Europe and Central Asia, 61 in the Americas, and 26 species in Africa **(Figure 4.47)**. There are no documented species causing positive impacts on good quality of life in Antarctica. This pattern translates to the number of documented impacts, with 46 per cent (241 impacts) of the positive impacts on good quality of life documented in Asia-Pacific, 21.6 per cent (113 impacts) in Europe and Central Asia, 15.6 per cent (82) in the Americas, and 8.4 per cent (44) in Africa **(Figure 4.48)**. Across all regions, good quality of life is mostly positively impacted through changes to material and immaterial assets (180 impacts). Health (111 impacts) is the second most positively impacted component of good quality of life across regions. There are 92 impacts on health in the Asia-Pacific region, 11 impacts in Africa, 6 impacts in Europe and Central Asia, and 2 impacts in the Americas. Social and cultural relationships is the second most impacted component to good quality of life, with 52 positive impacts in Asia-Pacific, 20 in the Americas, and 18 in Europe and Central Asia. Safety and freedom of choice and action are the two least positively impacted components of good quality of life across all regions.

Compared to negative impacts, fewer patterns emerge with positive impacts by taxonomic group and region. Plants are the dominant taxonomic group causing 37 per cent (213 impacts) of all positive impacts on good quality of life across most regions, including Europe and Central Asia, the Americas, and Africa. Of all documented positive impacts caused by invasive alien plants, 40 per cent are in the Asia-Pacific region (86 impacts). The remaining share of positive impacts occur in Europe and Central Asia (71 impacts; 33 per cent), the Americas (34 impacts; 16 per cent), and Africa (22 impacts; 10 per cent).

Invertebrates are responsible for 170 positive impacts, or 29.6 per cent of all positive impacts on good quality of life across regions. Invertebrates are the dominant taxonomic group positively affecting the Asia-Pacific region (108 impacts), accounting for 64 per cent of all invertebrate

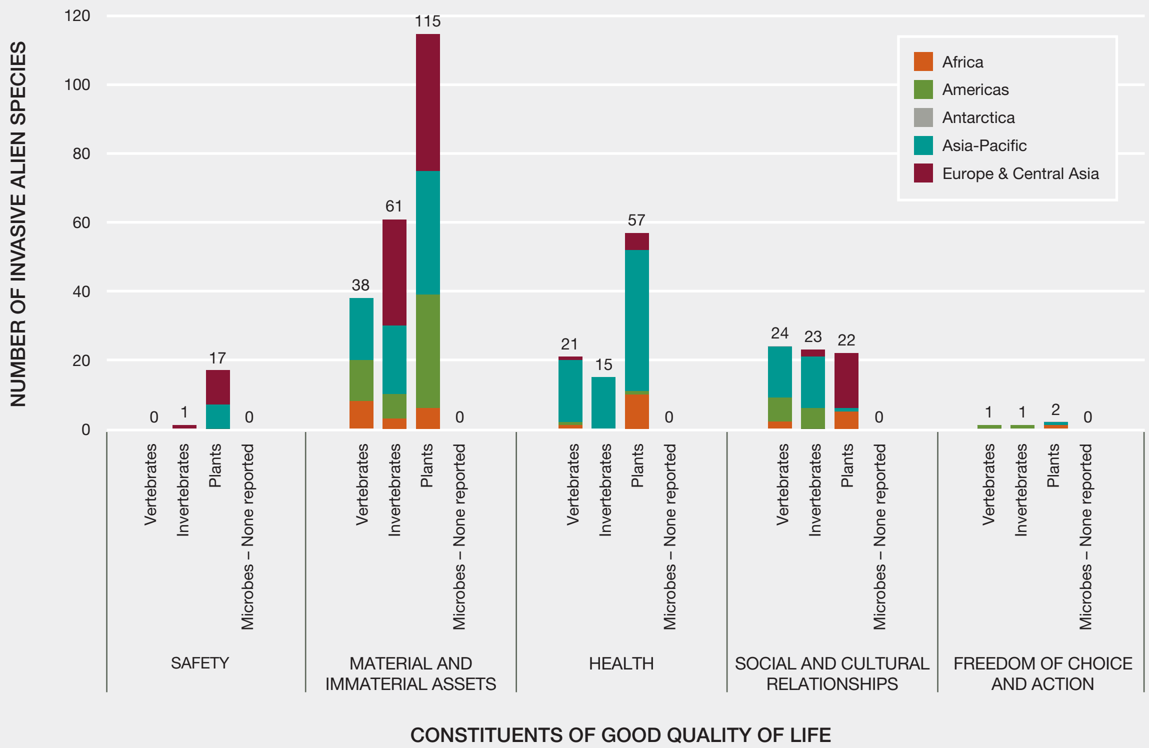


Figure 4 47 **Number of invasive alien species (y axis) causing positive impacts on constituents of good quality of life by taxonomic group and region (x axis).**

A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

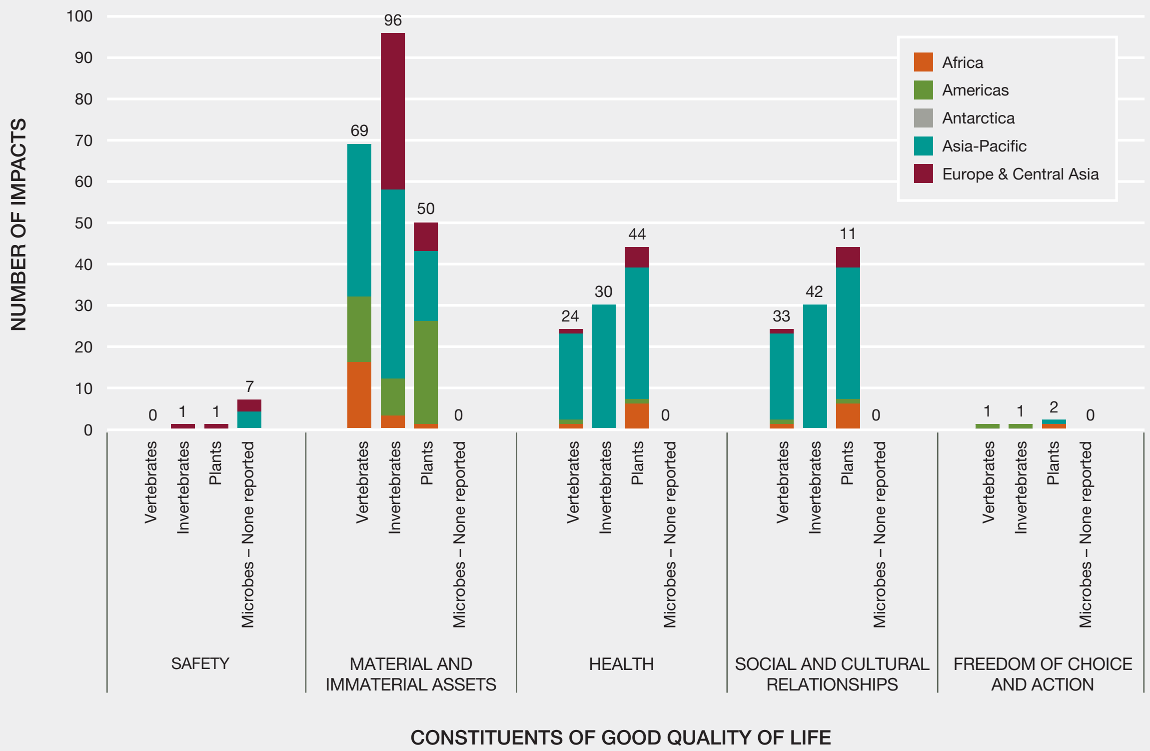


Figure 4 48 **Number of positive impacts (y axis) on constituents of good quality of life by taxonomic group and region (x axis).**

A data management report for the database of impacts developed through this chapter, with underlying data for this figure is available at https://doi.org/10.5281/zenodo.5766069

impacts. Europe and Central Asia account for 24 per cent of invertebrate impacts (41 impacts). The Americas (18 impacts; 10 per cent) and Africa (3 impacts; 2 per cent) are the regions with the fewest documented positive impacts from invertebrates.

Aside from microbes that do not have any documented positive impacts (**section 4.7.2**), vertebrates are the least documented taxonomic group causing positive impacts on good quality of life, accounting for 24 per cent of impacts by taxon. As with negative impacts, positive impacts caused by vertebrates are heavily concentrated in the Asia-Pacific region (77 impacts; 61 per cent of all positive impacts caused by vertebrates). The Americas document 30 positive impacts (24 per cent), Africa records 19 impacts (15 per cent) and Europe and Central Asia document only 1 impact (1 per cent) caused by invasive alien vertebrates on good quality of life.

4.6 REVIEW OF IMPACTS OF INVASIVE ALIEN SPECIES FOR INDIGENOUS PEOPLES AND LOCAL COMMUNITIES

This section presents the results of a systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species,[16] which supplements the chapter impact database. Of the 131 sources reviewed, a total of 124 sources provided evidence of impacts of invasive alien species on, or as perceived by Indigenous Peoples and local communities, with 79 sources containing direct information (e.g., from survey data, interviews and quotations) from Indigenous Peoples and local communities.

Overall, this review has revealed a total of 368 impacts on nature, nature's contributions to people and good quality of life, as documented by Indigenous Peoples and local communities. Indigenous Peoples and local communities

16. Data management report available at: https://doi.org/10.5281/zenodo.5760266

have identified a varied range of invasive alien species, with nearly two-thirds of the sources reviewed being associated with invasive alien plants. Additionally, they have documented the presence of invasive alien vertebrates and invertebrates, as well as a single microbe **(Table 4.30)**. This may represent a research bias towards invasive alien plants in rural communities, but all taxonomic groups are represented in this analysis.

Impacts are presented as they were described by Indigenous Peoples and local communities, but authors have assigned directions of impact following the classification used throughout this chapter (**section 4.1.2**). While it may seem straightforward to identify negative and positive impacts of alien species invasions on nature and native species (i.e., a native species suffers or is advantaged by the an invasive alien species) or on good quality of life (i.e., people derive a benefit from an invasive alien species), Indigenous Peoples and local communities have emphasized that a positive impact on nature's contribution to people or good quality of life may not be considered as wholly positive for their communities, and instead may represent the "least-worst" option (IPBES, 2022). For example, while the capture and sale invasive of alien fish species introduced into traditional fishing grounds may be considered as positive for some (Riedmiller, 1994; K. Smith *et al.*, 2010; Cid-Aguayo *et al.*, 2021), it may not be the case for Indigenous Peoples and local communities, especially when they have not had agency or choice in the initial introduction of the invasive alien species (K. Smith *et al.*, 2010; Broderstad & Eythórsson, 2014), and/or if their preference still is for the native species that have been displaced (IPBES, 2022). Therefore, when interpreting the findings of this review, it is important to consider the options available to Indigenous Peoples and local communities and whether their use of and adaptation to invasive alien species has been freely determined by choice. Some Indigenous Peoples and local communities lack resources, funding and capacity to voice and implement their preferences regarding management of biological invasions, and may have chosen eradication instead of adaptation if they had access to more resources (IPBES, 2022). However, it is also important to note that some Indigenous Peoples and local communities have shown considerable capacity for adaptation using their detailed and intimate knowledge and skills connected with their environment as well as partnerships with emerging technologies, and can be a model for resilience to future impacts (**Chapter 5**, **section 5.7**, Africa Uncensored, 2022; P. L. Howard, 2019). Overall, positive impacts documented on nature's contribution to people and good quality of life refer to where humans have derived a benefit, and yet they are often part of a more complex trade-off between positive and negative impacts inherent in socio-cultural-ecological systems.

4.6.1 Impacts on nature as documented by Indigenous Peoples and local communities

Indigenous Peoples and local communities report that approximately 92 per cent of impacts on nature caused by invasive alien species are negative, and only 8 per cent are positive impacts **(Table 4.31)**. They have observed an overall reduction in specific native species (31 per cent),

Table 4 30 **The number of Indigenous and local knowledge sources reviewed with information about the impacts of invasive alien plants, vertebrates, invertebrates and microbes.**

A data management report for this cross-chapter literature review on Indigenous Peoples and invasive alien species is available at: https://doi.org/10.5281/zenodo.5760266

Taxonomic group	Number and percentage of Indigenous and local knowledge sources reviewed	
Plants	80	65%
Invertebrates	14	11%
Vertebrates	19	15%
Microbe	1	1%
Multiple Taxa	10	8%
Total	**124**	**100%**

and a loss in vegetation cover and diversity due to invasive alien species (19 per cent), as well as negative impacts on native animals, including displacement, reduction in animal food and habitat and predation (7 per cent combined). Indigenous Peoples and local communities note that ecosystem processes, including fire regimes and regeneration, have also been disrupted by invasive alien species (e.g., Jevon & Shackleton, 2015), and that some invasive alien species are increasing the abundance of other invasive alien species. For example, local rice farmers in Cambodia report that the invasive shrub, *Mimosa pigra* (giant sensitive plant), has increased other invasive pests such as nematodes and rodents, which are more problematic for Indigenous Peoples and local communities in their rice fields (Rijal & Cochard, 2016).

Almost one-third of the reviewed sources highlight that invasive alien species have caused the reduction in specific native animal and plant species, with impacts occurring to species of similar niche or taxon (e.g., plants outcompeting other plants) or across different taxa (e.g., plants displacing fauna). For example, the Ifugao farmers in the Philippines have noted that *Pomacea canaliculata* (golden apple snail or "batikor" in local language) outcompetes native snails (R. C. Joshi *et al.*, 2001) and Aboriginal people in north-eastern Australia have reported that the invasion by *Rhinella marina* (cane toad) led to the disappearance of native frogs (Boll, 2006). The impact of invasive alien species on native species of a different taxon was highlighted by local communities in Nepal, who documented that the invasive vine, *Mikania micrantha* (bitter vine), limits food sources for

Table 4 31 **Number and type of impacts on nature caused by invasive alien species, as documented directly by Indigenous Peoples and local communities.**

A data management report for the systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species is available at: https://doi.org/10.5281/zenodo.5760266

Types of negative and positive impacts of invasive alien species on nature, as documented by Indigenous Peoples and local communities	Number of reports	Percentage of total reports
Total negative reports	57	92%
Reduced specific species	19	31%
Reduced vegetation diversity/abundance	12	19%
Limits regeneration	5	8%
Negative impact on biodiversity	4	6%
Altered fire regimes	3	5%
Physical damage to habitat	3	5%
Increased abundance of other invasive alien species	3	5%
Kills trees	3	5%
Displaces animals	2	3%
Kills fish	1	2%
Predation	1	2%
Reduced animal habitat/food	1	2%
Total positive reports	5	8%
Provided animal habitat/food	2	3%
Increased animals	1	2%
Assist regeneration by limiting grazing	1	2%
Increased vegetation abundance	1	2%

wildlife, resulting in large and potentially dangerous fauna (tigers, rhinos, boar) increasingly leaving the forest in search of food (Sullivan *et al.*, 2017). Indigenous Peoples and local communities value specific species that may be important to livelihoods, be totem or culturally important species, and indicator species for seasonal or environmental changes (Curran *et al.*, 2019; C. J. Robinson & Wallington, 2012).

Aside from specific species and ecological properties, Indigenous Peoples and local communities also report an overall negative impact on biodiversity (8 per cent of reports; **Table 4.31**) which reflects their understanding of the impacts on nature as a whole. Invasive alien species causing declines in biodiversity are seen as a degradation of the overall habitat (Sundaram *et al.*, 2012), or a reduction in the condition of the forest (Jevon & Shackleton, 2015), leading to a decline in the health of landscapes. For example, weeds have caused "significant upheaval to their Aboriginal ancestral landscapes" (Bach *et al.*, 2019).

Positive impacts of invasive alien species on nature represent less than 10 per cent of impacts reported by Indigenous Peoples and local communities. Most of the reported positive impacts concern the increases in vegetation structure and cover provided by larger invasive alien shrubs and trees **(Table 4.31)**. For example, in open grasslands or previously degraded landscapes, some invasive alien species have provided habitat structure or additional food for animals (Bach *et al.*, 2019), or assisted regeneration of native seedlings underneath spiky canopies as seedlings were protected from browsing by animals (R. T. Shackleton *et al.*, 2017).

Brazil provides an example of how Indigenous Peoples and local communities can experience a range of impacts on nature, including connections with different taxa and with ecosystem properties such as water regulation.

> "We live on an island surrounded by [invasive alien] acacia plants! Before, we hunted and fished, now we have bees that attack us and acacia plants that invade our farm plots as soon as we clear (burn) them, and they grow even stronger. I've killed rattlesnakes there that are attracted by the rats, and there have been more foxes and opossums, which damage the buriti palms. There are no more electric eels, and the water is rusty. You can't drink the water in the Manoá igarapé, and even our wells are drying up. The ingá trees have stopped producing fruit since the acacia appeared. Parrots used to make nests in São Domingo, but now the bees have taken over. Rolinha doves used to wake us up and tell us when it was going to rain; now those birds don't exist here anymore" (Souza *et al.*, 2018, p. 6)

4.6.2 Impacts on nature's contributions to people as documented by Indigenous Peoples and local communities

Of the 368 documented impacts of invasive alien species,[17] over 50 per cent are on nature's contributions to people, which reflects the direct connection and dependence of many Indigenous Peoples and local communities on nature's contributions to people for their livelihoods (R. T. Shackleton, Shackleton, *et al.*, 2019) and reveals they have valuable knowledge on more complex ecosystem processes and services (F. Walsh *et al.*, 2013; Ens, Pert, *et al.*, 2015). Traditional and customary practices have often been developed over a long period of time to respectfully derive services from nature (Sangha *et al.*, 2018). Although the number of documented impacts on nature's contributions to people were relatively balanced between negative (55 per cent) and positive (45 per cent) **(Table 4.32)**, the incidence of these impacts varied across categories. There are more negative than positive documented impacts on the provision of food and feed, on the availability and quality of water, and on cultural identities; whereas Indigenous Peoples and local communities report more positive than negative impacts on materials, labour and transport, energy, medicines, soil processes, physical and psychological experiences, and climate, with the last two categories mostly related to the provision of shade and ornamental aesthetics from plants **(Table 4.32)**.

For Indigenous Peoples and local communities, the provision of food and feed is the most negatively impacted (31 per cent) category of nature's contributions to people **(Table 4.32)**. This broad category includes the abundance and condition of wild food or crops for people, wild food and fodder for domestic animals and wildlife, as well as broader scale impacts such as a reduction in the size of land or interaction with other invasive alien species that cause crop damage. Impacts upon crops alone lead to various impacts on good quality of life, as local swidden farmers in West Africa documented in interviews that:

> "in decreasing order of importance, [*Imperata cylindrica* (cogon grass)], reduces crop yield, limits field size that family labour can handle, increases labour requirements for weeding, causes physical injury to the skin, reduces quality of tuber crops, increases the occurrence of bush fires in perennial crops, and increases the incidence of insects and pathogens of economic crops" (Chikoye *et al.*, 2000; Table 4, p. 485).

Many Indigenous Peoples and local communities highlighted the negative impact of invasive alien species on livestock

17. Data management report available at: https://doi.org/10.5281/zenodo.5760266

Table 4 32 **Number and type of impacts on nature's contributions to people caused by invasive alien species that were documented directly by Indigenous Peoples and local communities in peer-reviewed sources.**

A data management report for the systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species is available at: https://doi.org/10.5281/zenodo.5760266

Types of negative and positive impacts of invasive alien species on nature's contributions to people, as documented by Indigenous Peoples and local communities	Number of reports	Percentage of total reports
Total negative reports by category	103	55%
Food and feed (includes 13 reports (7%) of impacts on livestock health)	57	31%
Freshwater quantity	10	5%
Materials, companionship, labour	8	4%
Soils	7	4%
Supporting identities	6	3%
Freshwater quality	5	3%
Detrimental processes	2	1%
Maintenance of options	2	1%
Climate	1	1%
Energy	1	1%
Hazards	1	1%
Medicinal	1	1%
Physical/psychological experiences	1	1%
Pollination	1	1%
Total positive reports by category	84	45%
Food and Feed	22	12%
Energy	14	7%
Materials, companionship, labour	12	6%
Soils	11	6%
Medicinal	10	5%
Physical/psychological experiences	4	2%
Climate – shade	4	2%
Supporting identities	2	1%
Water quantity	2	1%
Air quality	1	1%
Habitat creation/maintenance	1	1%
Hazards	1	1%

health (7 per cent of reports), as a specific element within food and feed **(Table 4.32)**. These negative impacts subsequently affected their good quality of life, as poorer condition livestock need more labour to be looked after, and livestock have inherent cultural value. For example, Puri (2015) described how for local people from southern Karnataka, India, cattle are a cultural keystone species, and yet "*Lantana camara* [lantana] causes difficulties feeding cattle as it covers up and suppresses fodder grasses. This has led to underfed and malnourished animals, which has weakened them and led to increased vulnerability to disease, injury due to accidents, and attack by wild animals, such as leopards. People in these communities fear for their own safety – having to take cattle further into the forest, on to steeper and more marginal terrain, and having to stay longer every day." (Puri, 2015, p. 259).

Indigenous Peoples and local communities have reported significant impacts of invasive alien species on water resources, including water availability and security (5 per cent of reports) and water quality (3 per cent of reports). Their level of concern about these impacts was as high as that for impacts on livestock health **(Table 4.32)**. Indigenous Peoples and local communities have also documented negative impacts of invasive alien species on soils (4 per cent of reports), which include impacts on soil fertility, erosion, microbiological processes, and overall land degradation. In a similar holistic perspective to impacts on nature, Indigenous Peoples and local communities view soil health as connected to other ecosystem processes such as regulation of water and the provision of food and feed, health, and the land (Koichi *et al.*, 2012).

Indigenous Peoples and local communities have documented positive impacts of invasive alien species across multiple categories of nature's contributions to people, mostly on food and feed (12 per cent of reports), followed by energy (7 per cent), materials (6 per cent), soil processes (6 per cent), medicinal purposes (5 per cent), and physical/psychological experiences (2 per cent) and climate regulation (2 per cent), mostly related to shade and aesthetics from invasive trees **(Table 4.32)**. Positive impacts can generally be observed by Indigenous Peoples and local communities in two situations: where invasive alien species are introduced, recognized and used for a particular purpose, and where they have adapted to the invasive alien species in a way that is different or supplementary to the original purpose of introduction or unintentional introductions. Indigenous Peoples and local communities have however highlighted that the use of or adaptation to an invasive alien species may not always be their preferred option, while other Indigenous Peoples and local communities have shown capacity for adaptation (**section 4.6**).

There are many examples where Indigenous Peoples and local communities can derive food, energy, materials and recognize land rehabilitation from invasive alien species, in line with the original purpose of introduction. Invasive alien fish species including *Oncorhynchus tshawytscha* (Chinook salmon), *Lates niloticus* (Nile perch), *Cyprinus carpio* (common carp), and Tilapia species have been introduced to traditional waterways as a food resource, and several Indigenous Peoples and local communities use the invasive alien species in this way to sustain their livelihoods (Riedmiller, 1994; K. Smith *et al.*, 2010; Cid-Aguayo *et al.*, 2021). However, Indigenous Peoples and local communities are often not the agency in charge of such introductions (K. Smith *et al.*, 2010; Broderstad & Eythórsson, 2014), and, alongside use of the invasive alien species, they report negative impacts on the original food supply, such as native fish in this case (Macnaughton *et al.*, 2015; Santos & Nóbrega Alves, 2016; Cid-Aguayo *et al.*, 2021). Similarly, many invasive trees have been introduced for timber supply (e.g., *Acacia mearnsii* (black wattle)), as a fuel source for household energy (e.g., *Prosopis juliflora* (mesquite)), and for erosion control and land rehabilitation (e.g., *Grevillea banksii* (Banks' grevillea), *Prosopis juliflora*), and these species are used and recognized by Indigenous Peoples and local communities for these particular purposes (Duenn *et al.*, 2017; Kull *et al.*, 2019; C. M. Shackleton *et al.*, 2007). However, Indigenous Peoples and local communities report that whilst invasive alien species are used for these purposes, the materials or energy source may be of lower quality to the original native species that they have replaced (Kull *et al.*, 2019).

More commonly, Indigenous Peoples and local communities documented positive impacts where they adapted to the invasive alien species in new ways with additional or supplementary uses. For example, *Grevillea banksii* (Banks' grevillea) was introduced to Madagascar for erosion control but Indigenous Peoples and local communities now value this plant for honey production, as well as for charcoal and fuel, fencing, and as habitat for birds (Kull *et al.*, 2019). Branches of the invasive shrub, *Lantana camara* (lantana), are now used to make baskets for transporting goods, and supports basketry industry for local communities in southern India (Kannan *et al.*, 2014). Other adaptive uses for invasive alien plants include making manures and fertilizer, soaps, oils and glues, and in particular, adapting to use invasive alien plants as medicines (5 per cent of reports). Adaptation can lead to improvements in good quality of life, such as facilitating cultural knowledge transfer.

4.6.3 Impacts on good quality of life of Indigenous Peoples and local communities

Indigenous Peoples and local communities also experience impacts of invasive alien species on their good quality of life.[18] The systematic cross-chapter review on Indigenous

18. Data management report available at: https://doi.org/10.5281/zenodo.5760266

Peoples and local communities and invasive alien species highlights that over two-thirds of impacts on their good quality of life are negative (68 per cent), and less than one-third are positive (32 per cent) **(Table 4.33)**.

4.6.3.1 Affected constituents of good quality of life

When considering the different constituents of good quality of life (**Chapter 1**, **Table 1.4**; **section 4.1.2**; **Box 4.3**), Indigenous Peoples and local communities are experiencing both negative and positive impacts on material and immaterial assets, in a similar proportion, with 28 per cent and 24 per cent of all reports, respectively **(Table 4.33)**. However, when considering all the remaining elements of good quality of life, there are far more documented negative impacts than positive impacts of invasive alien species on human health (13 per cent negative, 1 per cent positive), safety (10 per cent negative, 1 per cent positive), and freedom of choice and action (8 per cent negative, no positive reports), and slightly more negative than positive reports for social, cultural and spiritual relationships (10 per cent negative and 7 per cent positive). Spiritual impacts may have been under-documented as, for many Indigenous Peoples and local communities, spirituality is a foundational consideration for all aspects of daily living and worldview, that is interconnected with more than one constituent of good quality of life (Robin *et al.*, 2022). However, spirituality may be private knowledge that is not shared in public research, or may be all encompassing and taken as an obvious component of everyday life that is therefore not singled out during interview questions (IPBES, 2022).

4.6.3.2 Themes directly documented from Indigenous Peoples and local communities

Indigenous Peoples and local communities have consistently identified themes within the literature that reviews how invasive alien species impact the five main constituents of their good quality of life **(Table 4.34)**. Some of these themes feed into multiple constituents of good quality of life, for example, maintaining access and mobility is considered by Indigenous Peoples and local communities in access to resources (material/immaterial assets, Adams *et al.*, 2018; Kent & Dorward, 2015), cultural sites (social/spiritual/cultural relationships; C. M. Shackleton *et al.*, 2007; Bach *et al.*, 2019) and the freedom to move as they have always done (freedom of choice or action, (Rettberg, 2010). This accounts for slightly different numbers of reports in **Table 4.33**, compared to **Table 4.34**.

Table 4 33 **Number of impacts of invasive alien species on the five constituents of good quality of life for Indigenous Peoples and local communities.**

A data management report for the systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species is available at: https://doi.org/10.5281/zenodo.5760266

Negative and positive impacts on the five constituents of good quality of life for Indigenous Peoples and local communities	Number of reports	Percentage of total reports
Total negative reports	81	68%
Material/Immaterial assets	33	28%
Health	15	13%
Safety	12	10%
Social/Spiritual/Cultural	12	10%
Freedom of choice/action	9	8%
Total positive reports	38	32%
Material/Immaterial assets	28	24%
Social/Spiritual/Cultural	8	7%
Health	1	1%
Safety	1	1%
Grand Total	**119**	

Table 4 34 **Impacts on good quality of life documented by Indigenous Peoples across different themes.**

Number and type of impacts on the good quality of life of Indigenous Peoples and local communities, by themes directly documented by Indigenous Peoples and local communities in the reviewed sources. Colours in the columns to the right indicate the constituents affected by the documented life theme. A data management report for the systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species is available at https://doi.org/10.5281/zenodo.5760266

Impacts on good quality of life across themes	Number of reports	Percentage of total reports	Affected constituent of good quality of life				
			Assets	Health	Safety	Relations	Freedom
Negative impacts							
Health	15	13%		■			
Labour – more difficult/costly/time/amount	14	11%	■				■
Access and Mobility	13	11%	■			■	■
Cultural knowledge transfer/practices/relations/values	9	7%				■	■
Safety	9	7%			■		
Livelihoods overall negatively impacted	7	6%	■	■	■	■	■
Abandon activities or land	4	3%	■				■
Damage to material assets	4	3%	■				
Reduced land area	4	3%	■				
Conflict individual level	3	2%			■	■	
Damage to cultural sites	3	2%				■	
Feeling disturbed by changes in environment and way of life	3	2%		■			
Affected industry/economy	2	2%	■				
Freedom of choice/action – considering future generations	2	2%					■
Enjoyment of areas	1	1%		■		■	
Reduced income	1	1%	■				
Increased expenditure	1	1%	■				
Reduced social cohesion/quality	1	1%				■	
Positive impacts							
Livelihood resource – income	9	7%	■				
Cultural knowledge transfer/practices/relations/values	5	4%		■		■	■
Develop an industry/employment	4	3%	■				
Labour is easier	2	2%	■				
Livelihood resource	2	2%	■				
Health	1	1%		■			
Livelihood resource – income savings	1	1%	■				
Livelihoods – housing	1	1%	■				
Relaxation	1	1%		■			

Impacts of invasive alien species on material and immaterial assets have been documented as negative and positive in similar proportions **(Table 4.33)**, but breaking this down into themes from Indigenous Peoples and local communities, it appears that positive impacts derive from gaining income or developing an industry (10 per cent of reports combined), and negative impacts translate into increased labour, reduced mobility and access, and less availability of traditional lands (28 per cent of reports combined) **(Table 4.34)**.

Some invasive alien species provide income streams and support Indigenous Peoples and local communities to engage in or develop an industry such as honey production, basketry, *Melaleuca* oil distilleries, sports fishing, hunting, or tourism (Kannan *et al.*, 2014; Aigo & Ladio, 2016; Ens, Fisher, *et al.*, 2015; Kull *et al.*, 2019; Maldonado Andrade, 2019; Fache, 2021). In some cases, local industries supports employment that maintains cultural connections, with long-lasting and broad benefits to health and good quality of life for Indigenous Peoples and local communities (A. Wright *et al.*, 2021). Industries based on invasive alien species can also provide a more stable income stream, such as charcoal-making, which is more reliable and as economically beneficial as rain-fed rice cultivation (Chandrasekaran & Swamy, 2016). Industries specialized on a single invasive alien species can however become a more susceptible income stream for people, and reduce the diversity of earlier income stream made before invasion, for example from a wide variety of non-timber forest products (Kannan *et al.*, 2014). A study on *Lantana camara* (lantana) in Karnataka, India, showed little difference in household income derived from invasive alien species compared to original forest resources (Kannan *et al.*, 2014). As noted before, while Indigenous Peoples and local communities can adapt to an invasive alien species and derived benefits, they may have preferred to maintain and protect the original native species, had this option been available (IPBES, 2022)

The positive reports of income from invasive alien species are contrasted with reports of harder labour, reduced access and mobility, abandoned traditional activities or abandoned/reduced land area **(Table 4.34)**. Reduced access and mobility, and increases in labour requirements due to invasive alien species were both equally documented by Indigenous Peoples and local communities (11 per cent of reports each). Invasive alien species can indeed reduce access to traditional lands, cultural sites or access to basic resources such as to clean water by physically blocking travelling routes, limiting mobility of people and making it more time consuming to reach resources, and even leading to the thought of traditional lands being "blocked" by invasive alien species (R. T. Shackleton *et al.*, 2017; Witt *et al.*, 2019). There were no reports of invasive alien species improving access and mobility for Indigenous Peoples and local communities, nor increasing the size of available land, and only one mention of an invasive forb, *Chromolaena odorata* (Siam weed), which made labour easier for some local rice farmers in Laos (Roder *et al.*, 1995). Ensuring the rights of Indigenous Peoples to maintain, use, and control their traditional lands (Article 26 of the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP)) is important for Indigenous Peoples and local communities to maintain their cultural identity and self-determination as well as be able to better respond to and manage biological invasions. Loss of access and rights to traditional lands has been highlighted as a driver of the establishment and spread of invasive alien species (IPBES, 2022; **Chapter 3**, **section 3.2.5**), which facilitates further negative impacts.

Health of Indigenous Peoples and local communities has been documented to be more negatively impacted (13 per cent of reports) than positively (1 per cent) **(Table 4.34)**. Negative health impacts include injury, allergies, toxicity, lack of access to clean water, but they have also been documented when the lands of Indigenous Peoples and local communities and nature were affected by invasive alien species (Sloane *et al.*, 2019), inducing stress and sadness from working on "sick country" (Maclean *et al.*, 2022), or feeling despair at the influence of humans in environmental change (Aigo & Ladio, 2016). Indirect impacts on health have also been documented such as from charcoal production derived from invasive alien species (Kull *et al.*, 2019), and there may be more indirect health effects that have not yet been documented in the literature. There are multiple ways by which health of Indigenous Peoples and local communities can be affected. For example, Rogers *et al.* (2017) document that for traditional Afar pastoralists in Ethiopia, *Prosopis* (mesquite) has indirectly reduced the availability of milk for domestic consumption and/or market, resulting in a lack of cash resources for education and healthcare. Afar pastoralists also observed that their economic status, social health, and community well-being are negatively affected, leading to reduced capacity to adapt to change and cope with environmental risks, as well as contributing to a widespread feeling of despair and uncertainty regarding their overall quality of life (Rogers *et al.*, 2017). Invasive alien species can also impact the safety and security of Afar pastoralists, as dense invasive alien plants can provide a hiding place for larger wildlife or criminals, causing violent conflict with Issa pastoralists over resources (Rogers *et al.*, 2017).

Impacts on society-wide good quality of life

There are 66 documented examples where invasive alien species have impacted the well-being of communities and societies at a higher level **(Table 4.35)**. More research with input from Indigenous Peoples and local communities is required on this topic as these society-level impacts

Table 4 35 **Number and type of impacts on society-wide good quality of life for Indigenous Peoples and local communities.**

A data management report for the systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species is available at https://doi.org/10.5281/zenodo.5760266

Negative and positive impacts of invasive alien species on society-wide good quality of life for Indigenous Peoples and local communities	Number of reports	Percentage of total reports
Total negative reports	55	83%
Conflict	15	23%
Cultural institutions	11	17%
Resource tenure	7	11%
Settlement/land-use	7	11%
Education/knowledge	6	9%
Governance	4	6%
Social stratification	4	6%
Social security	1	2%
Total positive reports	11	17%
Cultural institutions	3	5%
Education/knowledge	2	3%
Resource tenure	2	3%
Social stratification	2	3%
Governance	1	2%
Settlement/land use	1	2%
Grand Total	**66**	

have often been interpreted solely by authors of the publications. A vast majority (over 80 per cent) of these society-level impacts are negative, they include conflicts between groups, major changes in land use and resource tenure, and disruptions or other harms to ancestral cultural identities, laws and relationships (Amanor, 1991; Bekele *et al.*, 2018; Pretty Paint-Small, 2013; Costanza *et al.*, 2017; Sloane *et al.*, 2019). Some positive impacts have also been documented, highlighting that adaptation to invasive alien species can contribute, in some cases, to maintain cultural institutions and knowledge, and language transfer between generations, especially when Indigenous Peoples and local communities still have access to their traditional lands (Maldonado Andrade, 2019; Bach *et al.*, 2019). In some cases, invasive alien species have become part of the cultural identity of Indigenous Peoples and local communities (e.g., feral cattle in Hawaii, Fischer, 2007).

4.6.4 Indigenous Peoples and local communities: comparing positive and negative impacts of invasive alien species

The lack of data on the magnitude of impacts of invasive alien species on nature, nature's contributions to people and good quality of life, as assessed by Indigenous Peoples and local communities (**section 4.7.2**), poses a challenge in comparing impacts between studies, regions or communities. The magnitude of impacts of invasive alien species may not be simply categorized as either wholly positive or negative, as there are often trade-offs to be considered. In some cases, the positive impacts may be the "least-worst" option, while still having some negative effects (IPBES, 2022). In 12 of the reviewed sources,

Table 4 36 **Examples of invasive alien species with conflicting values.**

Sources retrieved from a systematic cross-chapter review on Indigenous Peoples and local communities and invasive alien species. Data management report available at https://doi.org/10.5281/zenodo.5760266

Comparison of beneficial and detrimental impacts within the text	Invasive alien species
"Only 37% of respondents said that mimosa could be (and sometimes was) used as firewood; 63% saw no plant uses" (Rijal & Cochard, 2016)	*Mimosa pigra* (giant sensitive plant)
"A fifth (20%) of respondents reported eating *O. stricta* fruit, with the remaining 80% saying they ate it only rarely or never. Significantly, more men reported eating *O. stricta* than women … Respondents mentioned that a lot of time and effort is needed to remove the small barbs (glochids) from the fruit and that it could only be eaten in moderation otherwise it would result in stomach 'irritation'" (R. T. Shackleton et al., 2017, p. 2433)	*Opuntia* stricta (erect prickly pear)
"Some people use the plant's milky latex sap as a livestock insecticide, applying it to insects that are attached to cattle. However, this is not widely practiced because, as a number of participants explained, the sap is also a skin irritant and will burn a person if any touches exposed skin." (Luizza et al., 2016)	*Cryptostegia grandiflora* (rubber vine)
"Respondents reported that they have other trees superior to *S. spectabilis* in their compounds that serve as shade, flower and fence provision, and wind brakes." (Mungatana & Ahimbisibwe, 2012, p. 189)	*Senna spectabilis* (whitebark senna)
"Health problems include animal teeth falling out from eating too many *P. juliflora* pods, as a pastoralist suggested, 'For two months of the year, the pods are good so animals can eat something; if they eat it every two days it creates no problems, but too much makes the teeth fall'" (Duenn et al., 2017, p. 571)	*Prosopis* ssp.
"Some farmers say Acheampong [*Chromolaena odoratum*] is bad. But if you are strong and can cut out its roots it is not bad. Maize grows well where Acheampong has been. It has some moisture in its roots and this is good. But if you can't cut out its roots it is trouble. It will grow back very quickly and spoil your crops" (Amanor, 1991, p. 9)	*Chromolaena odorata* (Siam weed)

Indigenous Peoples and local communities have conducted a comparison of the negative *versus* positive effects within the same study, considering these trade-offs. In 11 out of the 12 cases, invasive alien species were found to have more negative than positive impacts overall.

Particularly, Indigenous Peoples and local communities reported an equal number of negative and positive impacts on material and immaterial assets. Studies on the perspectives and experiences of Indigenous Peoples and local communities with invasive alien species have therefore sought further qualitative and contextual analysis through survey data to better understand these impacts **(Table 4.36)**.

4.6.5 Interactions between impacts and trends, drivers, management documented by Indigenous Peoples and local communities

Interaction of invasive alien species trends and impacts

Indigenous Peoples and local communities report changes in the impacts of invasive alien species depending on the trend in abundance over time. In this review, some impacts increased with time, whilst other impacts decreased depending on the interaction with livelihoods.[19] For example, *Paralithodes camtschaticus* (red king crab) was initially seen as a pest by the Saami fisher people in Norway, but was later viewed as a major economic resource (Broderstad & Eythórsson, 2014). In contrast, in Botswana, local people initially "embraced" *Prosopis juliflora* (mesquite), but as rates of spread increased in the 1990s, its negative impacts on livelihoods started to become a serious concern (Mosweu *et al.*, 2013).

Interactions of invasive alien species and other drivers of change amplify impacts for Indigenous Peoples and local communities

Other drivers were identified by Indigenous Peoples and local communities to interact with invasive alien species and amplify impacts. Climate-related drivers, including drought, rainfall and temperature variability were documented as reducing the resilience of livestock and crops to invasive alien species and diseases (Rettberg, 2010; Upadhyay *et al.*, 2020; Fenetahun *et al.*, 2020). A lack of resources, such as limited access to irrigation equipment or tools in the context of increased labour

19. Data management report available at: https://doi.org/10.5281/zenodo.5760266

demands, further put strain on Indigenous Peoples and local communities and their ability to cope with impacts of both invasive alien species and climate change (Rijal & Cochard, 2016). For some Indigenous Peoples and local communities, the introduction of invasive alien species to traditional lands and water is representative of human intervention by non-indigenous people at sacred landscapes, which causes additional distress on well-being due to historical and ongoing disempowerment (Aigo & Ladio, 2016; Bach *et al.*, 2019).

Impacts of interactions between management of invasive alien species by Indigenous Peoples and local communities

For Indigenous Peoples and local communities, positive impacts of invasive alien species can include opportunities for skills development, knowledge sharing and employment when managing biological invasions and controlling invasive alien species. For example, invasive weed management provided opportunities for elders to teach young Aboriginal peoples about culture, and to experiment with traditional burning regimes as a form of weed control (Bach & Larson, 2017). In North America, traditional methods to locate and harvest ash trees are being documented by Indigenous Nations in response to the spread of *Agrilus planipennis* (emerald ash borer), responsible for the death of culturally significant *Fraxinus nigra* (black ash). Family and tribal stories associated with traditional gathering areas are also being documented as part of this effort (Poland *et al.*, 2017; Reo *et al.*, 2017; **Box 4.14**).

Some reports have emphasized that the management of biological invasions can divert resources, such as time and money from other important priorities, or even cause harm itself. For instance, forest management committees in Nepal have been allocating a portion of their annual income for the management of *Mikania micrantha* (bitter vine), which would be otherwise spent on infrastructure development and social services (Sullivan & York, 2021). Additionally, they have reported an increase in the amount of labour and time required for controlling invasive alien species **(Table 4.34)**. As mentioned in previous sections, Indigenous Peoples and local communities have also reported experiencing health problems when controlling invasive alien species, particularly through the excessive or unsafe use of pesticides (Head & Atchison, 2015; Machekano *et al.*, 2017).

4.7 DISCUSSION AND FUTURE DIRECTIONS

4.7.1 Models and scenarios of impacts

Authors have conducted a systematic literature review of 778 papers on models and scenarios involving biological invasions, of which 171 papers address the impacts of invasive alien species.[20] Most studies consider the impacts of invasive alien species to native species or native ecosystems, and 18 per cent (31 papers) consider the impacts of invasive alien species on nature's contributions to people, 8 per cent (14 papers) on good quality of life, and 1.8 per cent (3 papers) on Indigenous and local knowledge **(Table 4.37)**. Material contributions (assets) are the most well-studied, both in nature's contributions to people and good quality of life. Most model and scenario studies on impacts of invasive alien species are conducted in the Americas, followed by Europe and Central Asia, and Asia and the Pacific **(Table 4.38)**. The United States has been the most extensively modelled country, followed by Australia, New Zealand, and Canada **(Table 4.39)**. Modelling studies in Europe often include more than one country and these studies are covered under the "several" category in **Table 4.39**. Process-based models are the most frequently used (81 papers) to study the impacts of invasive alien species, followed by correlative models (68 papers), hybrids (19 papers) and expert-based system (3 papers). The largest proportion of studies used exploratory scenario (138 papers), followed by policy-screening scenario (17 papers) and target-seeking scenario (16 papers). Climate change is the largest scenario type (60 papers), followed by invasive alien species managements. The most modelled taxonomic group for impact assessment is invertebrates (63 papers), followed by plants (54 papers), mammals (23 papers), and fish (22 papers). Terrestrial realm is the most frequently modelled realm (126 papers) followed by inland waters and marine.

Models quantifying the impacts of invasive alien species can be a helpful tool to inform decision-makers and stakeholders as they evaluate management options. The systematic review showed that a large proportion of model and scenarios studies focus on predicting the potential distribution ranges of invasive alien species (61 per cent), often using climate change scenarios (48 per cent), but the efforts to evaluate their impacts on nature's contributions to people, good quality of life and Indigenous local knowledge are limited. Building such models faces numerous challenges (Venette, 2015; Leung *et al.*, 2012) because the impacts of invasive alien species on nature, nature's

20. Data management report available at https://doi.org/10.5281/zenodo.5706520

contributions to people, and good quality of life are complex and highly context-dependent, and differ among invaded regions (Essl *et al.*, 2020; Kumschick *et al.*, 2015). Moreover, predicting future trajectories of the impacts of invasive alien species depends on the development of reliable scenarios for the introduction, time lags, and spread of the invasive alien species, but such attempts are still limited (Corrales *et al.*, 2018; Essl *et al.*, 2019). Currently, predicted trajectories of invasive alien species are primarily based on experts' knowledge and opinions from western regions, and inputs from other regions are rare (Essl *et al.*, 2020). Recently, however, conceptual frameworks for building alien species scenarios are emerging (Lenzner *et al.*, 2019), and future predictions of invasive alien species incursions and spread have been evaluated at the continental scale (Seebens *et al.*, 2021). Those studies will help to develop scenario-based assessments, such as climate change (IPCC, 2014) or biodiversity loss (IPBES, 2016), for biological invasions in the near future. Moreover, standardized global impact assessment schemes (Bacher *et al.*, 2018; IUCN, 2020; Vimercati *et al.*, 2022) and databases, such as InvaCost for the economic costs of biological invasions on a global scale (Diagne, Leroy, *et al.*, 2020), are available. A recent InvaCost study showed rising economic costs of biological invasions both in management and damage caused by invasive alien species (Diagne, Leroy, *et al.*, 2020; Diagne, Turbelin, *et al.*, 2021). Although there is no such global database nor study for the impacts of invasive alien species on native species or native ecosystems (but see **section 4.3.1**), it is most likely that those impacts are also increasing, since the number of invasive alien species establishments is still increasing globally (Seebens *et al.*, 2017; **Chapter 2**, **section 2.2.1**). Combining the predicted distribution of invasive alien species with those studies will provide an excellent opportunity to estimate the impacts of invasive alien species in a changing world.

The systematic literature review on scenarios and models completed for this assessment only focused on studies published in English, resulting in a potential bias towards western countries, especially English-speaking countries. Indeed, the United States is by far the most represented country in the dataset (23 per cent), followed by Australia (8 per cent), New Zealand (3 per cent) and Canada (3 per cent). Countries in other regions, especially Africa, are much less prevalent or missing altogether A recent study showed that non-English studies can contribute to improve our knowledge in conservation biology (T. Amano *et al.*, 2021), as well as estimation of the costs of biological invasions (Angulo, Diagne, *et al.*, 2021).

Table 4 37 **Number of publications on the impacts of invasive alien species on nature's contributions to people, good quality of life, and Indigenous and local knowledge, using models and scenarios.**

Data management report available at https://doi.org/10.5281/zenodo.5706520

	Type of impact	Both	Negative	Positive	Total
Nature's contributions to people					
No		16	120	4	140
Yes		7	22	2	31
	Material	1	12	1	14
	Non-material		3		3
	Regulating	6	7	1	14
Good quality of life					
No		22	129	6	157
Yes		1	13		14
	Material	1	8		9
	Non-material		5		5
Indigenous and local knowledge					
No		23	139	6	168
Yes			3		3

Table 4 38 **Number of publications per region on the impacts of invasive alien species using models and scenarios.**

Data management report available at https://doi.org/10.5281/zenodo.5706520

IPBES regions	Number of papers
The Americas	73
Europe and Central Asia	41
Asia and the Pacific	30
NA/NS (Not applicable/Not stated)	10
Africa; The Americas; Asia and the Pacific; Europe and Central Asia	8
Africa	4
Africa; Europe and Central Asia	1
Asia and the Pacific; The Americas	1
The Americas; Africa	1
The Americas; Asia and the Pacific; Africa	1
The Americas; Asia and the Pacific; Europe and Central Asia	1
Grand Total	**171**

Table 4 39 **Number of publications per country (top 12) on the impacts of invasive alien species using models and scenarios.**

Data management report available at https://doi.org/10.5281/zenodo.5706520

Countries	Number of papers
United States of America (the)	54
Several	37
Australia	11
New Zealand	7
Canada	6
France	4
Mexico	4
Finland	3
Germany	3
Italy	3
Japan	3
Portugal	3

4.7.2 Challenges for future studies of impacts (based on knowledge gaps)

Chapter 4 identifies a number of challenges that may limit the understating of impacts of invasive alien species. This section highlights the main challenges that have been identified in the hope that future research will help close these important knowledge gaps. Aiming for a more complete and global understanding of the impact of invasions will contribute to their successful management and governance (Nuñez *et al.*, 2020; **Chapters 5** and **6**).

The data and information presented in this chapter reveal substantial geographical and taxonomical gaps on the documentation, quantification and understanding of impacts, with lesser-studied regions potentially more affected, and lesser-studied taxa potentially more impactful (e.g., invasive alien viruses, bacteria, protists, fungi). The quality and quantity of impact information available for different taxa, units of analysis, regions and realms differ greatly, and research efforts for invasive alien species impacts are unevenly distributed geographically, temporally, and taxonomically.

The impact database developed through this chapter highlights the incompleteness of information on impacts of invasive alien species in Central Asia (mainly due to language barriers) and Africa. There are also discernible biases within regions. For example, in Africa, most impacts are documented from South Africa; eastern and northern Africa being much less covered.

These biases are observed across all realms, but especially in marine ecosystems, where the extent and timing of research efforts lag behind terrestrial studies (Ojaveer *et al.*, 2015). Quantitative data on ecological impacts are generally scarce, even in well-studied regions. Although research on marine invasive alien species is relatively recent (initiated in the 1960s and 1970s), there are already distinct geographic and taxonomic knowledge biases on impacts of marine invasive alien species. Impacts for the vast majority of marine alien species have not been quantitatively or experimentally studied over sufficiently long temporal and spatial scales, and their cumulative and synergetic connections with other drivers of change affecting the marine environment are largely unknown. A literature survey on alien marine macroalgae revealed information on impacts for only 30 species globally (Davidson *et al.*, 2015). Evidence for most of the documented ecosystem impacts in European seas is based on expert judgement or correlations, with only 13 per cent of the documented impacts inferred from manipulative or natural experiments. A similar paucity of impact data is apparent in North America. A recent synthesis of global ecological impacts[21] comprises 76 species, about 4 per cent of documented marine alien species, and the ecological impacts of 49 of the species were quantified in only one study each.

This chapter also highlights biases in the study of impacts of invasive alien species across units of analysis: in the marine realm, most studies were confined to intertidal/shallow subtidal areas, and in the terrestrial realm few impacts have been documented in deserts, tundra and high elevation mountainous habitats.

The impact database developed through this chapter also reveals a lack of understanding and synthesis of impacts of invasive alien microbes across all regions of the world. Some microbes are pathogens of plants, animals or humans, and due to their small size and parasitic lifestyle, many microbes can frequently be transported, introduced and established. While microbes can be considered as invasive alien organisms (Nuñez *et al.*, 2020; H. E. Roy *et al.*, 2017), they have been long ignored in the field of ecology, and this could be a reason for their small representation.

Similar to trends in publications in other disciplines (Nuñez *et al.*, 2021), many of the publications reviewed in this chapter focus on impacts occurring in a narrow set of wealthy countries. Although references in other languages could drastically improve the understanding of impacts of invasive alien species, about 95 per cent of the publications listed in the impact databased developed through this chapter are in English, severely underrepresenting studies in non-English scientific journals (Angulo, Diagne, *et al.*, 2021; Nuñez & Amano, 2021).

The intrusion of geopolitical boundaries in biological invasion science constitutes another information-related challenge, as invasive alien species are often transported from one region to another within the same country. Subsequently, a species native to one region may, under certain definitions, be considered invasive in another region in the same country, especially in large countries (Nelufule *et al.*, 2022). In the impact database developed through this chapter, geopolitical boundaries have been considered, i.e., species were only defined alien if they crossed national borders.

Context dependency presents a fundamental challenge (Sapsford *et al.*, 2020) when determining whether impacts are deemed detrimental or beneficial. Assessing the directionality of impact can be influenced by subjective human perceptions and values, resulting in potential disagreement among different stakeholders. Some invasive alien species have conflicting values associated with them, whereby they may cause negative impacts for some, but may be treasured by others. They may negatively affect some native taxa, but create conditions that favour other native taxa (Vitule *et al.*, 2012) or have economic benefits to some sectors **(Box 4.10)**. Impacts may also change over time, with some species having very low negative

21. https://www.marinespecies.org/introduced/

impacts for long periods of time, before they become highly problematic (**Chapter 1**, **section 1.4.4**; **Chapter 2**, **section 2.2.2**; Essl *et al.*, 2012). Furthermore, the same invasive alien species can also have a large impact in one area but no impact in another (Zenni & Nuñez, 2013). A deeper understanding on the socioecological context of conflict species and time lags will contribute to more successful management programmes (**Chapter 5**, **section 5.6.1.2**).

This chapter highlights several other research and knowledge gaps that impede a comprehensive understanding of impacts of invasive alien species. Compared to the information available on impacts on nature, there is incomplete data on impacts on nature's contributions to people and good quality of life. Furthermore, there is very little systematic research on gender differences in impacts of invasive alien species beyond anecdotal evidence of direct impacts (for further examples see IPBES, 2022). Most studies on impacts of marine invasive alien species relate to impacts on nature, including ecosystem health. The number of marine invasive alien species with sufficient data to satisfy the criteria for "significant negative impact" is small, as the understanding of marine ecosystem functions is constrained. Unless impacts are conspicuous, induce direct economic cost, or impinge on human health, they fail to elicit public awareness, attract funding, or result in scientific analysis (Katsanevakis *et al.*, 2014; Ruiz *et al.*, 1999). Improving the data and understanding on the extent and variety of the impacts marine invasive alien species create, singly and cumulatively, will contribute to providing timely and efficient management and policy instruments.

Finally, impacts resulting from interactions amongst invasive alien species and with other drivers of change, are largely misunderstood. Interactions among co-occurring invasive alien species ("invasional meltdown"; **Glossary**; **Chapter 1**, **section 1.3.4**; **Chapter 3**, **section 3.3.5.1**; Simberloff & Von Holle, 1999) or with other drivers of change can exacerbate their impacts and facilitate additional invasive alien species, increasing competition with native species, and creating new challenges for restoration (**Glossary**) of native habitats (Kuebbing & Nuñez, 2016). For instance, global extinctions **(Box 4.4)** are often caused by multiple factors, including invasive alien species. Understanding the interactions of invasive alien species with other drivers of change such as land- and sea-use change, climate change, pollution and sociocultural drivers (e.g., hunting of wildlife), will improve the understanding of impacts of invasive alien species and inform future predictions of the impact of invasive alien species.

REFERENCES

Adams, V. M., Douglas, M. M., Jackson, S. E., Scheepers, K., Kool, J. T., & Setterfield, S. A. (2018). Conserving biodiversity and Indigenous bush tucker: Practical application of the strategic foresight framework to invasive alien species management planning. *Conservation Letters*, *11*(4), e12441. https://doi.org/10.1111/conl.12441

Adelino, J. R. P., Heringer, G., Diagne, C., Courchamp, F., Faria, L. D. B., & Zenni, R. D. (2021). The economic costs of biological invasions in Brazil: A first assessment. *NeoBiota*, *67*, 349–374. https://doi.org/10.3897/neobiota.67.59185

Africa Uncensored (Director). (2022). *Fighting Laikipia's Prickly Pear*. https://www.youtube.com/watch?v=S-X7nsF7t6Y

Aigo, J., & Ladio, A. (2016). Traditional Mapuche ecological knowledge in Patagonia, Argentina: Fishes and other living beings inhabiting continental waters, as a reflection of processes of change. *Journal of Ethnobiology and Ethnomedicine*, *12*(1), 56. https://doi.org/10.1186/s13002-016-0130-y

Akashi, Y., & Mueller-Dombois, D. (1995). A landscape perspective of the Hawaiian rain forest dieback. *Journal of Vegetation Science*, *6*(4), 449–464. https://doi.org/10.2307/3236343

Albins, M. A. (2015). Invasive Pacific lionfish *Pterois volitans* reduce abundance and species richness of native Bahamian coral-reef fishes. *Marine Ecology Progress Series*, *522*, 231–243. https://doi.org/10.3354/meps11159

Albins, M. A., & Hixon, M. A. (2008). Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. *Marine Ecology Progress Series*, *367*, 233–238. https://doi.org/10.3354/meps07620

Albrecht, G., McMahon, C. R., Bowman, D. M. J. S., & Bradshaw, C. J. A. (2009). Convergence of Culture, Ecology, and Ethics: Management of Feral Swamp Buffalo in Northern Australia. *Journal of Agricultural and Environmental Ethics*, *22*(4), 361–378. https://doi.org/10.1007/s10806-009-9158-5

Al-Frayh, A., Hasnain, S. M., Gad-el-Rab, M. O., Al-Turki, T., Al-Mobeireek, K., & Al-Sedairy, S. T. (1999). Human Sensitization to *Prosopis juliflora* Antigen in Saudi Arabia. *Annals of Saudi Medicine*, *19*(4), 331–336. https://doi.org/10.5144/0256-4947.1999.331

Allen, M. S. (1984). A review of archaeobotany and palaeoethnobotany in Hawaii. *Hawaiian Archaeology*, *1*(1), 19–30. https://hilo.hawaii.edu/maunakea/library/reference.php?view=977

Allen, M. S. (1989). Archaeobotanical assemblages from the Anahulu rockshelters. In P. V. Kirch (Ed.), *Prehistoric Hawaiian Occupation in the Anahulu Valley, Oahu Island: Excavations in Three Inland Rockshelters* (pp. 83–102). University of California. https://escholarship.org/uc/item/375247r9

Aloo, P. A., Njiru, J., Balirwa, J. S., & Nyamweya, C. S. (2017). Impacts of Nile Perch, *Lates niloticus*, introduction on the ecology, economy and conservation of Lake Victoria, East Africa. *Lakes & Reservoirs: Science, Policy and Management for Sustainable Use*, *22*(4), 320–333. https://doi.org/10.1111/lre.12192

Alves, J. D. S., Fabricante, J. R., Reis, L. B. de O., De Moraes, G. S., & Silva, E. K. C. (2018). Biological invasion by *Cenchrus ciliaris* L.: Is there an impact on Caatinga composition and diversity of herbaceous stratum? *Revista de Biologia Neotropical*, *14*(2), 101–110. https://doi.org/10.5216/rbn.v14i2.41717

Amano, N., Bankoff, G., Findley, D. M., Barretto-Tesoro, G., & Roberts, P. (2021). Archaeological and historical insights into the ecological impacts of pre-colonial and colonial introductions into the Philippine Archipelago. *The Holocene*, *31*(2), 313–330. https://doi.org/10.1177/0959683620941152

Amano, T., Berdejo-Espinola, V., Christie, A. P., Willott, K., Akasaka, M., Báldi, A., Berthinussen, A., Bertolino, S., Bladon, A. J., Chen, M., Choi, C.-Y., Kharrat, M. B. D., Oliveira, L. G. de, Farhat, P., Golivets, M., Aranzamendi, N. H., Jantke, K., Kajzer-Bonk, J., Aytekin, M. Ç. K., … Sutherland, W. J. (2021). Tapping into non-English-language science for the conservation of global biodiversity. *PLoS Biology*, *19*(10), e3001296. https://doi.org/10.1371/journal.pbio.3001296

Amanor, K. S. (1991). Managing the Fallow: Weeding Technology and Environmental Knowledge in the Krobo District of Ghana. *Agriculture and Human Values*, *8*(1–2), 5–13. https://doi.org/10.1007/BF01579651

Amaya-Villarreal, A. M., & Miguel Renjifo, L. (2010). Effects of Gorse (*Ulex europaeus*) on the birds of a high Andean forest edge. *Ornitologia Colombiana*, *10*, 11–25. https://asociacioncolombianadeornitologia.org/ojs/index.php/roc/article/view/232

Anagnostakis, S. L. (1987). Chestnut Blight: The Classical Problem of an Introduced Pathogen. *Mycologia*, *79*(1), 23–37. https://doi.org/10.2307/3807741

Andres, J. A., Thampy, P. R., Mathieson, M. T., Loye, J., Zalucki, M. P., Dingle, H., & Carroll, S. P. (2013). Hybridization and adaptation to introduced balloon vines in an Australian soapberry bug. *Molecular Ecology*, *22*(24), 6116–6130. https://doi.org/10.1111/mec.12553

Angienda, P. O., Lee, H. J., Elmer, K. R., Abila, R., Waindi, E. N., & Meyer, A. (2011). Genetic structure and gene flow in an endangered native tilapia fish (*Oreochromis esculentus*) compared to invasive Nile tilapia (*Oreochromis niloticus*) in Yala swamp, East Africa. *Conservation Genetics*, *12*(1), 243–255. https://doi.org/10.1007/s10592-010-0136-2

Angulo, E., Ballesteros-Mejia, L., Novoa, A., Duboscq-Carra, V. G., Diagne, C., & Courchamp, F. (2021). Economic costs of invasive alien species in Spain. *NeoBiota*, *67*, 267–297. https://doi.org/10.3897/neobiota.67.59181

Angulo, E., Diagne, C., Ballesteros-Mejia, L., Adamjy, T., Ahmed, D. A., Akulov, E., Banerjee, A. K., Capinha, C., Dia, C. A. K. M., Dobigny, G., Duboscq-Carra, V. G., Golivets, M., Haubrock, P. J., Heringer, G., Kirichenko, N., Kourantidou, M., Liu, C., Nuñez, M. A., Renault, D., … Courchamp, F. (2021). Non-English languages enrich scientific knowledge: The example of economic costs of biological invasions. *Science of The Total Environment*, *775*, 144441. https://doi.org/10.1016/j.scitotenv.2020.144441

Angulo, E., Hoffmann, B. D., Ballesteros-Mejia, L., Taheri, A., Balzani, P., Bang, A., Renault, D., Cordonnier, M., Bellard, C., Diagne, C., Ahmed, D. A., Watari, Y., & Courchamp, F. (2022). Economic costs of invasive alien ants worldwide. *Biological Invasions*, *24*(7), 2041–2060. https://doi.org/10.1007/s10530-022-02791-w

Antolić, B., Žuljević, A., Despalatović, M., Grubelić, I., & Cvitkovic, I. (2008). Impact of the invasive green alga *Caulerpa racemosa* var. *Cylindracea* on the epiphytic macroalgal assemblage of *Posidonia oceanica* seagrass rhizomes in the Adriatic Sea. *Nova Hedwigia*, *86*(1–2), 155–167. https://doi.org/10.1127/0029-5035/2008/0086-0155

Antsulevich, A., & Välipakka, P. (2000). *Cercopagis pengoi*—New important food object of the Baltic herring in the Gulf of Finland. *International Review of Hydrobiology: A Journal Covering All Aspects of Limnology and Marine Biology*, *85*(5-6), 609–619. https://doi.org/10.1002/1522-2632(200011)85:5/6<609::AID-IROH609>3.0.CO;2-S

Aravind, N. A., Rao, D., Ganeshaiah, K. N., Shaanker, R. U., & Poulsen, J. G. (2010). Impact of the invasive plant, *Lantana camara*, on bird assemblages at Malé Mahadeshwara Reserve Forest, South India. *Tropical Ecology*, *51*(2S), 325–338. https://hdl.handle.net/10568/20465

Asare-Nuamah, P. (2022). Smallholder farmers' adaptation strategies for the management of fall armyworm (*Spodoptera frugiperda*) in rural Ghana. *International Journal of Pest Management*, *68*(1), 8–18. https://doi.org/10.1080/09670874.2020.1787552

Ashe, D., & Driscoll, T. (2013). *B.A. Steinhagen Reservoir—2013 Survey Report* (B.A. Steinhagen Reservoir, p. 27). Inland Fisheries Division. https://tpwd.texas.gov/publications/pwdpubs/lake_survey/pwd_rp_t3200_1376/2013.phtml#download

Aslan, C. E., & Dickson, B. G. (2020). Non-native plants exert strong but under-studied influence on fire dynamics. *NeoBiota*, *61*, 47–64. https://doi.org/10.3897/neobiota.61.51141

Aslan, C. E., Liang, C. T., Galindo, B., Kimberly, H., & Topete, W. (2016). The Role of Honey Bees as Pollinators in Natural Areas. *Natural Areas Journal*, *36*(4), 478–488. https://doi.org/10.3375/043.036.0413

Atala, C., Pertierra, L. R., Aragón, P., Carrasco-Urra, F., Lavín, P., Gallardo-Cerda, J., Ricote-Martínez, N., Torres-Díaz, C., & Molina-Montenegro, M. A. (2019). Positive interactions among native and invasive vascular plants in Antarctica: Assessing the "nurse effect" at different spatial scales. *Biological Invasions*, *21*(9), 2819–2836. https://doi.org/10.1007/s10530-019-02016-7

Athanassiou, C. G., Phillips, T. W., & Wakil, W. (2019). Biology and Control of the Khapra Beetle, *Trogoderma granarium*, a Major Quarantine Threat to Global Food Security. *Annual Review of Entomology*, *64*, 131–148. https://doi.org/10.1146/annurev-ento-011118-111804

Athens, J. S. (1997). Hawaiian Native Lowland Vegetation in Prehistory. In P. V. Kirch & T. L. Hunt (Eds.), *Historical Ecology in the Pacific Islands: Prehistoric Environmental and Landscape Change* (pp. 248–270). Yale University Press. https://doi.org/10.2307/j.ctt211qz1v

Athens, J. S. (2009). *Rattus exulans* and the catastrophic disappearance of Hawai'i's native lowland forest. *Biological Invasions*, *11*(7), 1489–1501. https://doi.org/10.1007/s10530-008-9402-3

Athni, T. S., Shocket, M. S., Couper, L. I., Nova, N., Caldwell, I. R., Caldwell, J. M., Childress, J. N., Childs, M. L., De Leo, G. A., Kirk, D. G., MacDonald, A. J., Olivarius, K., Pickel, D. G., Roberts, S. O., Winokur, O. C., Young, H. S., Cheng, J., Grant, E. A., Kurzner, P. M., … Mordecai, E. A. (2021). The influence of vector-borne disease on human history: Socio-ecological mechanisms. *Ecology Letters*, *24*(4), 829–846. https://doi.org/10.1111/ele.13675

Atyosi, Z., Ramarumo, L. J., & Maroyi, A. (2019). Alien Plants in the Eastern Cape Province in South Africa: Perceptions of Their Contributions to Livelihoods of Local Communities. *Sustainability*, *11*(18), 5043. https://doi.org/10.3390/su11185043

Aukema, J. E., Leung, B., Kovacs, K., Chivers, C., Britton, K. O., Englin, J., Frankel, S. J., Haight, R. G., Holmes, T. P., Liebhold, A. M., McCullough, D. G., & Von Holle, B. (2011). Economic Impacts of Non-Native Forest Insects in the Continental United States. *PLoS ONE*, *6*(9), e24587. https://doi.org/10.1371/journal.pone.0024587

Aydin, M., Düzgüneş, E., & Karadurmuş, U. (2016). Rapa whelk (*Rapana venosa* Valenciennes, 1846) fishery along the Turkish coast of the Black Sea. *Journal of Aquaculture Engineering and Fisheries Research*, 85–96. https://doi.org/10.3153/JAEFR16011

Bach, T. M., Kull, C. A., & Rangan, H. (2019). From killing lists to healthy country: Aboriginal approaches to weed control in the Kimberley, Western Australia. *Journal of Environmental Management*, *229*, 182–192. https://doi.org/10.1016/j.jenvman.2018.06.050

Bach, T. M., & Larson, B. M. H. (2017). Speaking About Weeds: Indigenous Elders' Metaphors for Invasive Species and Their Management. *Environmental Values*, *26*(5), 561–581. https://doi.org/10.3197/096327117X15002190708119

Bacheler, N. M., Schobernd, C. M., Harter, S. L., David, A. W., Sedberry, G. R., & Kellison, G. T. (2022). Reef fish community structure along the southeastern US Atlantic continental shelf break and upper slope appears resistant to increasing lionfish (*Pterois volitans/miles*) density. *Bulletin of Marine Science*, *98*(1), 75–98. https://doi.org/10.5343/bms.2021.0008

Bacher, S., Blackburn, T. M., Essl, F., Genovesi, P., Heikkilä, J., Jeschke, J. M., Jones, G., Keller, R., Kenis, M., Kueffer, C., Martinou, A. F., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Richardson, D. M., Roy, H. E., Saul, W.-C., Scalera, R., … Kumschick, S. (2018). Socio-economic impact classification of alien taxa (SEICAT). *Methods in Ecology and Evolution*, *9*(1), 159–168. https://doi.org/10.1111/2041-210X.12844

Bailey, G. A. (2018). *Architecture and Urbanism in the French Atlantic Empire: State, Church, and Society, 1604-1830* (Vol. 1). McGill-Queen's University Press. https://books.google.co.jp/books?id=onleDwAAQBAJ

Baird, H. P., Janion-Scheepers, C., Stevens, M. I., Leihy, R. I., & Chown, S. L. (2019). The ecological biogeography of indigenous and introduced Antarctic springtails. *Journal of Biogeography*, *46*(9), 1959–1973. https://doi.org/10.1111/jbi.13639

Baldacconi, R., & Corriero, G. (2009). Effects of the spread of the alga *Caulerpa racemosa* var. *Cylindracea* on the sponge assemblage from coralligenous concretions of the Apulian coast (Ionian Sea, Italy). *Marine Ecology*, *30*(3), 337–345. https://doi.org/10.1111/j.1439-0485.2009.00282.x

Bálint, T., Ferenczy, J., Kátai, F., Kiss, I., Kráczer, L., Kufcsák, O., Láng, G., Polyhos, C., Szabó, I., Szegletes, T., & Nemcsók, J. (1997). Similarities and differences between the massive eel (*Anguilla anguilla* L.) devastations that occurred in Lake Balaton in 1991 and 1995. *Ecotoxicology and Environmental Safety*, *37*(1), 17–23. https://doi.org/10.1006/eesa.1996.1509

Balirwa, J. S. (2017). *Lake Victoria wetlands and the ecology of the Nile Tilapia, Oreochromis Niloticus Linné*. Routledge. https://doi.org/10.4324/9780203749630

Balirwa, J. S., Chapman, C. A., Chapman, L. J., Cowx, I. G., Geheb, K., Kaufman, L., Lowe-McConnell, R. H., Seehausen, O., Wanink, J. H., Welcomme, R. L., & Witte, F. (2003). Biodiversity and Fishery Sustainability in the Lake Victoria Basin: An Unexpected Marriage? *BioScience*, *53*(8), 703–715. https://doi.org/10.1641/0006-3568(2003)053[0703:BAFSIT]2.0.CO;2

Ballesteros-Mejia, L., Angulo, E., Diagne, C., Cooke, B., Nuñez, M. A., & Courchamp, F. (2021). Economic costs of biological invasions in Ecuador: The importance of the Galapagos Islands. *NeoBiota*, *67*, 375–400. https://doi.org/10.3897/neobiota.67.59116

Ballew, N. G., Bacheler, N. M., Kellison, G. T., & Schueller, A. M. (2016). Invasive lionfish reduce native fish abundance on a regional scale. *Scientific Reports*, *6*(1), 32169. https://doi.org/10.1038/srep32169

Bang, A., Cuthbert, R. N., Haubrock, P. J., Fernandez, R. D., Moodley, D., Diagne, C., Turbelin, A. J., Renault, D., Dalu, T., & Courchamp, F. (2022). Massive economic costs of biological invasions despite widespread knowledge gaps: A dual setback for India. *Biological Invasions*, *24*(7), 2017–2039. https://doi.org/10.1007/s10530-022-02780-z

Banks, P. B., Newsome, A. E., & Dickman, C. R. (2000). Predation by red foxes limits recruitment in populations of eastern grey kangaroos. *Austral Ecology*, *25*(3), 283–291. https://doi.org/10.1046/j.1442-9993.2000.01039.x

Bardosh, K. L., Ryan, S. J., Ebi, K., Welburn, S., & Singer, B. (2017). Addressing vulnerability, building resilience: Community-based adaptation to vector-borne diseases in the context of global change. *Infectious Diseases of Poverty*, *6*(1), 166. https://doi.org/10.1186/s40249-017-0375-2

Bariw, S. A., Kudadze, S., & Adzawla, W. (2020). Prevalence, effects and management of fall army worm in the Nkoranza South Municipality, Bono East region of Ghana. *Cogent Food & Agriculture*, *6*(1), 1800239. https://doi.org/10.1080/23311932.2020.1800239

Barnes, I., Fourie, A., Wingfield, M. J., Harrington, T. C., McNew, D. L., Sugiyama, L. S., Luiz, B. C., Heller, W. P., & Keith, L. M. (2018). New *Ceratocystis* species associated with rapid death of *Metrosideros polymorpha* in Hawai'i. *Persoonia : Molecular Phylogeny and Evolution of Fungi*, *40*, 154–181. https://doi.org/10.3767/persoonia.2018.40.07

Barratt, J., Chan, D., Sandaradura, I., Malik, R., Spielman, D., Lee, R., Marriott, D., Harkness, J., Ellis, J., & Stark, D. (2016). *Angiostrongylus cantonensis*: A review of its distribution, molecular biology and clinical significance as a human pathogen. *Parasitology*, *143*(9), 1087–1118. https://doi.org/10.1017/S0031182016000652

Barrios, J., Bolaños, J., & López, R. (2007). Blanqueamiento de arrecifes coralinos por la invasión de *Kappaphycus alvarezii* (rhodophyta) en isla cubagua, estado nueva esparta, Venezuela. *Boletín Del Instituto Oceanográfico de Venezuela*, *46*(2), 147–152. https://repositorioslatinoamericanos.uchile.cl/handle/2250/227223

Bartlett, D., Milliken, S., & Parmar, D. (2018). '*Prosopis* for Prosperity': Using an invasive non-native shrub to benefit rural livelihoods in India. *Current Science*, *114*(10), 2142. https://doi.org/10.18520/cs/v114/i10/2142-2146

Bartlett, J. C., Convey, P., Pertierra, L. R., & Hayward, S. A. (2020). An insect invasion of Antarctica: The past, present and future distribution of *Eretmoptera murphyi* (Diptera, Chironomidae) on Signy Island. *Insect Conservation and Diversity*, *13*(1), 77–90. https://doi.org/10.1111/icad.12389

Batish, D. R., Kohli, R. K., Singh, H. P., & Kaur, G. (2012). Biology, Ecology and Spread of the Invasive Weed *Parthenium hysterophorus* in India. *Invasive Alien Plants An Ecological Appraisal for the Indian Subcontinent*, 10–18. https://doi.org/10.1079/9781845939076.0010

Baudron, F., Zaman-Allah, M. A., Chaipa, I., Chari, N., & Chinwada, P. (2019). Understanding the factors influencing fall armyworm (*Spodoptera frugiperda* J.E. Smith) damage in African smallholder maize fields and quantifying its impact on yield. A case study in Eastern Zimbabwe. *Crop Protection*, *120*, 141–150. https://doi.org/10.1016/j.cropro.2019.01.028

Beardmore, B. (2015). Boater Perceptions of Environmental Issues Affecting Lakes in Northern Wisconsin. *Journal of the American Water Resources Association (JAWRA)*, *51*(2), 537–549. https://doi.org/10.1111/jawr.12265

Becking, L. E., Bussel, T. van, Engel, M. S., Christianen, M. J. A., & Debrot, A. O. (2014). *Proximate response of fish, conch, and sea turtles to the presence of the invasive seagrass Halophila stipulacea in Bonaire*. Institute for Marine Resources & Ecosystem Studies (IMARES). https://www.wur.nl/en/Publication-details.htm?publicationId=publication-way-343536383735

Bédry, R., de Haro, L., Bentur, Y., Senechal, N., & Galil, B. S. (2021). Toxicological risks on the human health of populations living around the Mediterranean Sea linked to the invasion of non-indigenous marine species from the Red Sea: A review. *Toxicon*, *191*, 69–82. https://doi.org/10.1016/j.toxicon.2020.12.012

Beever, E. A., Simberloff, D., Crowley, S. L., Al-Chokhachy, R., Jackson, H. A., & Petersen, S. L. (2019). Social–ecological mismatches create conservation challenges in introduced species management. *Frontiers in Ecology and the Environment*, *17*(2), 117–125. https://doi.org/10.1002/fee.2000

Bekele, K., Haji, J., Legesse, B., Shiferaw, H., & Schaffner, U. (2018). Impacts of woody invasive alien plant species on rural livelihood: Generalized propensity score evidence from *Prosopis* spp. invasion in Afar Region in Ethiopia. *Pastoralism: Research, Policy and Practice*, *8*(1), 28. https://doi.org/10.1186/s13570-018-0124-6

Beköz, A. B., Beköz, S., Yilmaz, E., Tüzün, S., & Beköz, Ü. (2013). Consequences of the increasing prevalence of the poisonous *Lagocephalus sceleratus* in southern Turkey. *Emergency Medicine Journal*, *30*(11), 954–955. https://doi.org/10.1136/emermed-2011-200407

Belgrad, B. A., & Griffen, B. D. (2015). Rhizocephalan infection modifies host food consumption by reducing host activity levels. *Journal of Experimental Marine Biology and Ecology*, *466*, 70–75. https://doi.org/10.1016/j.jembe.2015.02.011

Bellard, C., Marino, C., & Courchamp, F. (2022). Ranking threats to biodiversity and why it doesn't matter. *Nature Communications*, *13*(1), 2616. https://doi.org/10.1038/s41467-022-30339-y

Ben Souissi, J., Rifi, M., Ghanem, R., Ghozzi, L., Boughedir, W., & Azzurro, E. (2014). *Lagocephalus sceleratus* (Gmelin, 1789) expands through the African coasts towards the Western Mediterranean Sea: A call for awareness. *Management of Biological Invasions*, *5*(4), 357–362. https://doi.org/10.3391/mbi.2014.5.4.06

Benkwitt, C. E. (2016). Central-place foraging and ecological effects of an invasive predator across multiple habitats. *Ecology*, *97*(10), 2729–2739. https://www.jstor.org/stable/44081850

Bergstrom, D. M. (2021). Maintaining Antarctica's isolation from non-native species. *Trends in Ecology & Evolution*, *37*(1), 5–9. https://doi.org/10.1016/j.tree.2021.10.002

Bergstrom, D. M., Lucieer, A., Kiefer, K., Wasley, J., Belbin, L., Pedersen, T. K., & Chown, S. L. (2009). Indirect effects of invasive species removal devastate World Heritage Island. *Journal of Applied Ecology*, *46*(1), 73–81. https://doi.org/10.1111/j.1365-2664.2008.01601.x

Berthon, K. (2015). How do native species respond to invaders? Mechanistic and trait-based perspectives. *Biological Invasions*, *17*, 2199–2211. http://dx.doi.org/10.1007/s10530-015-0874-7

Bezerra-Santos, M. A., Mendoza-Roldan, J. A., Thompson, R. C. A., Dantas-Torres, F., & Otranto, D. (2021). Illegal Wildlife Trade: A Gateway to Zoonotic Infectious Diseases. *Trends in Parasitology*, *37*(3), 181–184. https://doi.org/10.1016/j.pt.2020.12.005

Bhattacharyya, J., & Larson, B. M. H. (2014). The Need for Indigenous Voices in Discourse about Introduced Species: Insights from a Controversy over Wild Horses. *Environmental Values*, *23*(6), 663–684. https://doi.org/10.3197/096327114X13947900181031

Bhattacharyya, J., Slocombe, D. S., & Murphy, S. D. (2011). The "Wild" or "Feral" Distraction: Effects of Cultural Understandings on Management Controversy Over Free-Ranging Horses (*Equus ferus caballus*). *Human Ecology*, *39*(5), 613–625. https://doi.org/10.1007/s10745-011-9416-9

Blackburn, T. M., Essl, F., Evans, T., Hulme, P. E., Jeschke, J. M., Kühn, I., Kumschick, S., Marková, Z., Mrugala, A., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D. M., Sendek, A., Vilà, M., Wilson, J. R. U., Winter, M., … Bacher, S. (2014). A Unified Classification of Alien Species Based on the Magnitude of their Environmental Impacts. *PLoS Biology*, *12*(5), e1001850. https://doi.org/10.1371/journal.pbio.1001850

Blanchard, M. (2009). Recent expansion of the slipper limpet population (*Crepidula fornicata*) in the Bay of Mont-Saint-Michel (Western Channel, France). *Aquatic Living Resources*, *22*(1), 11–19. https://doi.org/10.1051/alr/2009004

Blanchet, E., Boudreault, L., Bernard, D., Campbell, G., Therrien, D., Lachapelle-Gill, I., & Pinceloup, N. (2022). *Assemblée citoyenne sur l'avenir du frêne noir et de la vannerie: Ce que nous avons entendus* (p. 45 pp). Bureau du Ndakina.

Bloomer, J. P., & Bester, M. N. (1991). Effects of hunting on population characteristics of feral cats on Marion Island. *South African Journal of Wildlife Research*, *21*(4), 97-102. https://hdl.handle.net/10520/EJC116884

Bodey, T. W., Carter, Z. T., Haubrock, P. J., Cuthbert, R. N., Welsh, M. J., Diagne, C., & Courchamp, F. (2022). Building a synthesis of economic costs of biological invasions in New Zealand. *PeerJ*, *10*, e13580. https://doi.org/10.7717/peerj.13580

Böhm, M., Dewhurst-Richman, N. I., Seddon, M., Ledger, S. E. H., Albrecht, C., Allen, D., Bogan, A. E., Cordeiro, J., Cummings, K. S., Cuttelod, A., Darrigran, G., Darwall, W., Fehér, Z., Gibson, C., Graf, D. L., Köhler, F., Lopes-Lima, M., Pastorino, G., Perez, K. E., … Collen, B. (2021). The conservation status of the world's freshwater molluscs. *Hydrobiologia*, *848*(12–13), 3231–3254. https://doi.org/10.1007/s10750-020-04385-w

Boll, V. (2006). Following *Garkman*, the frog, in North Eastern Arnhem Land (Australia). *Australian Zoologist*, *33*(4), 436–445. https://doi.org/10.7882/AZ.2006.016

Bolpagni, R. (2021). Towards global dominance of invasive alien plants in freshwater ecosystems: The dawn of the Exocene? *Hydrobiologia*, *848*(9), 2259–2279. https://doi.org/10.1007/s10750-020-04490-w

Bonebrake, T. C., Guo, F., Dingle, C., Baker, D. M., Kitching, R. L., & Ashton, L. A. (2019). Integrating Proximal and Horizon Threats to Biodiversity for Conservation. *Trends in Ecology & Evolution*, *34*(9), 781–788. https://doi.org/10.1016/j.tree.2019.04.001

Bonney, S., Andersen, A., & Schlesinger, C. (2017). Biodiversity impacts of an invasive grass: Ant community responses to *Cenchrus ciliaris* in arid Australia. *Biological Invasions*, *19*(1), 57–72. https://doi.org/10.1007/s10530-016-1263-6

Boone, R. B., Conant, R. T., Sircely, J., Thornton, P. K., & Herrero, M. (2018). Climate change impacts on selected global rangeland ecosystem services. *Global Change Biology*, *24*(3), 1382–1393. https://doi.org/10.1111/gcb.13995

Bortolus, A., Carlton, J. T., & Schwindt, E. (2015). Reimagining South American coasts: Unveiling the hidden invasion history of an iconic ecological engineer. *Diversity and Distributions*, *21*(11), 1267–1283. https://doi.org/10.1111/ddi.12377

Boylen, C. W., Eichler, L. W., & Madsen, J. D. (1999). Loss of native aquatic plant species in a community dominated by Eurasian watermilfoil. *Hydrobiologia*, *415*, 207–211. https://doi.org/10.1023/A:1003804612998

Braby, C. E., & Somero, G. N. (2006). Ecological gradients and relative abundance of native (*Mytilus trossulus*) and invasive (*Mytilus galloprovincialis*) blue mussels in the California hybrid zone. *Marine Biology*, *148*(6), 1249–1262. https://doi.org/10.1007/s00227-005-0177-0

Bradshaw, C. J. A., Hoskins, A. J., Haubrock, P. J., Cuthbert, R. N., Diagne, C., Leroy, B., Andrews, L., Page, B., Cassey, P., Sheppard, A. W., & Courchamp, F. (2021). Detailed assessment of the reported economic costs of invasive species in Australia. *NeoBiota*, *67*, 511–550. https://doi.org/10.3897/neobiota.67.58834

Bradshaw, C. J. A., Leroy, B., Bellard, C., Roiz, D., Albert, C., Fournier, A., Barbet-Massin, M., Salles, J.-M., Simard, F., & Courchamp, F. (2016). Massive yet grossly underestimated global costs of invasive insects. *Nature Communications*, *7*(1), 12986. https://doi.org/10.1038/ncomms12986

Branch, G. M., Odendaal, F., & Robinson, T. B. (2010). Competition and facilitation between the alien mussel *Mytilus galloprovincialis* and indigenous species: Moderation by wave action. *Journal of Experimental Marine Biology and Ecology*, *383*(1), 65–78. https://doi.org/10.1016/j.jembe.2009.10.007

Branch, G. M., & Steffani, C. N. (2004). Can we predict the effects of alien species? A case-history of the invasion of South Africa by *Mytilus galloprovincialis* (Lamarck). *Journal of Experimental Marine Biology and Ecology*, *300*(1–2), 189–215. https://doi.org/10.1016/j.jembe.2003.12.007

Brannock, P. M., & Hilbish, T. J. (2010). Hybridization results in high levels of sterility and restricted introgression between invasive and endemic marine blue mussels. *Marine Ecology Progress Series*, *406*, 161–171. https://doi.org/10.3354/meps08522

Bregnbak, D., Menné, T., & Johansen, J. D. (2015). Airborne contact dermatitis caused by common ivy (*Hedera helix* L. ssp. Helix). *Contact Dermatitis*, *72*(4), 243–244. https://doi.org/10.1111/cod.12337

Broderstad, E. G., & Eythórsson, E. (2014). Resilient communities? Collapse and recovery of a social-ecological system in Arctic Norway. *Ecology and Society*, *19*(3), 1. https://doi.org/10.5751/ES-06533-190301

Brodier, S., Pisanu, B., Villers, A., Pettex, E., Lioret, M., Chapuis, J.-L., & Bretagnolle, V. (2011). Responses of seabirds to the rabbit eradication on Ile Verte, sub-Antarctic Kerguelen Archipelago. *Animal Conservation*, *14*(5), 459–465. https://doi.org/10.1111/j.1469-1795.2011.00455.x

Brooke, M. de L., Bonnaud, E., Dilley, B. J., Flint, E. N., Holmes, N. D., Jones, H. P., Provost, P., Rocamora, G., Ryan, P. G., Surman, C., & Buxton, R. T. (2018). Seabird population changes following mammal eradications on islands. *Animal Conservation*, *21*(1), 3–12. https://doi.org/10.1111/acv.12344

Brothers, C. A., & Blakeslee, A. M. H. (2021). Alien vs predator play hide and seek: How habitat complexity alters parasite mediated host survival. *Journal of Experimental Marine Biology and Ecology*, *535*, 151488. https://doi.org/10.1016/j.jembe.2020.151488

Brown, G. P., Phillips, B. L., & Shine, R. (2011). The ecological impact of invasive cane toads on tropical snakes: Field data do not support laboratory-based predictions. *Ecology*, *92*(2), 422–431. https://doi.org/10.1890/10-0536.1

Brugueras, S., Fernández-Martínez, B., Martínez-de la Puente, J., Figuerola, J., Porro, T. M., Rius, C., Larrauri, A., & Gomez-Barroso, D. (2020). Environmental drivers, climate change and emergent diseases transmitted by mosquitoes and their vectors in southern Europe: A systematic review. *Environmental Research*, *191*, 110038. https://doi.org/10.1016/j.envres.2020.110038

Brundu, G. (2015). Plant invaders in European and Mediterranean inland waters: Profiles, distribution, and threats. *Hydrobiologia*, *746*(1), 61–79. https://doi.org/10.1007/s10750-014-1910-9

Bruzzese, E., & McFadyen, R. (2006). Arrival of leaf-feeding willow sawfly *Nematus oligospilus* Förster in Australia – pest or beneficial? *Plant Protection Quarterly*, *21*(1), 43–44. https://caws.org.nz/PPQ202122/PPQ%2021-1%20pp043-44%20Bruzzese.pdf

Bryson, C. T., Maddox, V. L., & Carter, R. (2008). Spread of Cuban club-rush (*Oxycaryum cubense*) in the southeastern United States. *Invasive Plant Science and Management*, *1*(3), 326–329. https://doi.org/10.1614/IPSM-08-083.1

Bullard, S. G., Lambert, G., Carman, M. R., Byrnes, J., Whitlatch, R. B., Ruiz, G., Miller, R. J., Harris, L., Valentine, P. C., Collie, J. S., Pederson, J., McNaught, D. C., Cohen, A. N., Asch, R. G., Dijkstra, J., & Heinonen, K. (2007). The colonial ascidian *Didemnum* sp. A: Current distribution, basic biology and potential threat to marine communities of the northeast and west coasts of North America. *Journal of Experimental Marine Biology and Ecology*, *342*(1), 99–108. https://doi.org/10.1016/j.jembe.2006.10.020

Bulleri, F., Balata, D., Bertocci, I., Tamburello, L., & Benedetti-Cecchi, L. (2010). The seaweed *Caulerpa racemosa* on Mediterranean rocky reefs: From passenger to driver of ecological change. *Ecology*, *91*(8), 2205–2212. https://doi.org/10.1890/09-1857.1

Bulleri, F., & Piazzi, L. (2015). Variations in importance and intensity of competition underpin context dependency in the effects of an invasive seaweed on resident assemblages. *Marine Biology*, *162*(2), 485–489. https://doi.org/10.1007/s00227-014-2563-y

Burrowes, P. A., & De la Riva, I. (2017). Unraveling the historical prevalence of the invasive chytrid fungus in the Bolivian Andes: Implications in recent amphibian declines. *Biological Invasions*, *19*(6), 1781–1794. https://doi.org/10.1007/s10530-017-1390-8

Butchart, S. H. M. (2008). Red List Indices to measure the sustainability of species use and impacts of invasive alien species. *Bird Conservation International*, *18*(S1), S245–S262. https://doi.org/10.1017/S095927090800035X

Butchart, S. H. M., Stattersfield, A. J., & Collar, N. J. (2006). How many bird extinctions have we prevented? *Oryx*, *40*(3), 266–278. https://doi.org/10.1017/S0030605306000950

Bye, R. A. (1981). Quelites-ethnoecology of edible greens-past, present and future. *Journal of Ethnobiology*, *1*(1), 109–123. https://ethnobiology.org/sites/default/files/pdfs/JoE/1-1/Bye1981.pdf

Cabin, R. J. (2013). *Restoring Paradise: Rethinking and Rebuilding Nature in Hawaii* (Illustrated edition). Latitude 20, University of Hawai'i Press.

Cadée, G. C. (2001). Herring gulls learn to feed on a recent invader in the Dutch Wadden Sea, the Pacific oyster *Crassostrea gigas*. *Basteria*, *65*(1/3), 33–42. https://natuurtijdschriften.nl/pub/597203

Cambray, J. A. (2003). Impact on indigenous species biodiversity caused by the globalisation of alien recreational freshwater fisheries. *Hydrobiologia*, *500*(1), 217–230. https://doi.org/10.1023/A:1024648719995

Canavan, S., Kumschick, S., Le Roux, J. J., Richardson, D. M., & Wilson, J. R. U. (2019). Does origin determine environmental impacts? Not for bamboos. *Plants, People, Planet*, *1*(2), 119–128. https://doi.org/10.1002/ppp3.5

Canonico, G. C., Arthington, A., McCrary, J. K., & Thieme, M. L. (2005). The effects of introduced tilapias on native biodiversity. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *15*(5), 463–483. https://doi.org/10.1002/aqc.699

Capizzi, D., Monaco, A., Genovesi, P., Scalera, R., & Carnevali, L. (2018). Impact of alien mammals on human health. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health*. CABI International.

Cárdenas, L., Leclerc, J.-C., Bruning, P., Garrido, I., Détrée, C., Figueroa, A., Astorga, M., Navarro, J. M., Johnson, L. E., Carlton, J. T., & Pardo, L. (2020). First mussel settlement observed in Antarctica reveals the potential for future invasions. *Scientific Reports*, *10*(1), 5552. https://doi.org/10.1038/s41598-020-62340-0

Carlsen, K. M., & Weismann, K. (2007). Phytophotodermatitis in 19 children admitted to hospital and their differential diagnoses: Child abuse and herpes simplex virus infection. *Journal of the American Academy of Dermatology*, *57*(5), S88–S91. https://doi.org/10.1016/j.jaad.2006.08.034

Carlton, J. T., Keith, I., & Ruiz, G. M. (2019). Assessing marine bioinvasions in the Galápagos Islands: Implications for conservation biology and marine protected areas. *Aquatic Invasions*, *14*(1), 1–20. https://doi.org/10.3391/ai.2019.14.1.01

Carnegie, A. J., & Pegg, G. S. (2018). Lessons from the Incursion of Myrtle Rust in Australia. *Annual Review of Phytopathology*, *56*(1), 457–478. https://doi.org/10.1146/annurev-phyto-080516-035256

Caro, A. U., Guiñez, R., Ortiz, V., & Castilla, J. C. (2011). Competition between a native mussel and a non-indigenous invader for primary space on intertidal rocky shores in Chile. *Marine Ecology Progress Series*, *428*, 177–185. https://doi.org/10.3354/meps09069

Caron, V., Ede, F., Sunnucks, P., & O'Dowd, D. J. (2014). Distribution and rapid range expansion of the introduced willow sawfly *Nematus oligospilus* Förster (Hymenoptera: Tenthredinidae) in Australasia. *Austral Entomology*, *53*(2), 175–182. https://doi.org/10.1111/aen.12067

Carpio, A. J., Guerrero-Casado, J., Barasona, J. A., Tortosa, F. S., Vicente, J., Hillström, L., & Delibes-Mateos, M. (2017). Hunting as a source of alien species: A European review. *Biological Invasions*, *19*(4), 1197–1211. https://doi.org/10.1007/s10530-016-1313-0

Carroll, S. P. (2007). Natives adapting to invasive species: Ecology, genes, and the sustainability of conservation. *Ecological Research*, *22*(6), 892–901. https://doi.org/10.1007/s11284-007-0352-5

Carroll, S. P., & Boyd, C. (1992). Host Race Radiation in the soapberry bug: Natural history with the history. *Evolution*, *46*(4), 1052–1069. https://doi.org/10.1111/j.1558-5646.1992.tb00619.x

Carroll, S. P., & Loye, J. E. (2012). Soapberry Bug (Hemiptera: Rhopalidae: Serinethinae) Native and Introduced Host Plants: Biogeographic Background of Anthropogenic Evolution. *Annals of the Entomological Society of America*, *105*(5), 671–684. https://doi.org/10.1603/AN11173

Carroll, S. P., Loye, J. E., Dingle, H., Mathieson, M., Famula, T. R., & Zalucki, M. P. (2005). And the beak shall inherit – evolution in response to invasion. *Ecology Letters*, *8*(9), 944–951. https://doi.org/10.1111/j.1461-0248.2005.00800.x

Castellanos-Galindo, G. A., Robertson, D. R., Pacheco-Chaves, B., Angulo, A., & Chong-Montenegro, C. (2019). Atlantic Tarpon in the Tropical Eastern Pacific 80 years after it first crossed the Panama Canal. *Reviews in Fish Biology and Fisheries*, *29*(2), 401–416. https://doi.org/10.1007/s11160-019-09565-z

Castilla, J. C., Guiñez, R., Caro, A. U., & Ortiz, V. (2004). Invasion of a rocky intertidal shore by the tunicate *Pyura praeputialis* in the Bay of Antofagasta, Chile. *Proceedings of the National Academy of Sciences*, *101*(23), 8517–8524. https://doi.org/10.1073/pnas.0401921101

Castro-Díez, P., Alonso, Á., Saldaña-López, A., & Granda, E. (2021). Effects of widespread non-native trees on regulating ecosystem services. *Science of The Total Environment*, *778*, 146141. https://doi.org/10.1016/j.scitotenv.2021.146141

Castro-Díez, P., Vaz, A. S., Silva, J. S., van Loo, M., Alonso, Á., Aponte, C., Bayón, Á., Bellingham, P. J., Chiuffo, M. C., DiManno, N., Julian, K., Kandert, S., La Porta, N., Marchante, H., Maule, H. G., Mayfield, M. M., Metcalfe, D., Monteverdi, M. C., Núñez, M. A., … Godoy, O. (2019). Global effects of non-native tree species on multiple ecosystem services. *Biological Reviews*, *94*(4), 1477–1501. https://doi.org/10.1111/brv.12511

Catenazzi, A., Lehr, E., Rodriguez, L. O., & Vredenburg, V. T. (2011). *Batrachochytrium dendrobatidis* and the collapse of anuran species richness and abundance in the upper Manu National Park, southeastern Peru. *Conservation Biology: The Journal of the Society for Conservation Biology*, *25*(2), 382–391. https://doi.org/10.1111/j.1523-1739.2010.01604.x

Cavicchioli, R. (2015). Microbial ecology of Antarctic aquatic systems. *Nature Reviews Microbiology*, *13*, 691–706. https://doi.org/10.1038/nrmicro3549

Cebrian, E., Linares, C., Marschal, C., & Garrabou, J. (2012). Exploring the effects of invasive algae on the persistence of gorgonian populations. *Biological Invasions*, *14*(12), 2647–2656. https://doi.org/10.1007/s10530-012-0261-6

Čerevková, A., Bobuľská, L., Miklisová, D., & Renčo, M. (2019). A case study of soil food web components affected by *Fallopia japonica* (Polygonaceae) in three natural habitats in Central Europe. *Journal of Nematology*, *51*, e2019–e2042. https://doi.org/10.21307/jofnem-2019-042

Chandrasekaran, S., Nagendran, N. A., Pandiaraja, D., Krishnankutty, N., & Kamalakannan, B. (2008). Bioinvasion of *Kappaphycus alvarezii* on corals in the Gulf of Mannar, India. *Current Science*, *94*(9), 1167–1172. https://www.jstor.org/stable/24100697

Chandrasekaran, S., & Swamy, P. S. (2016). Ecological and Socioeconomic Impacts of *Prosopis juliflora* Invasion in the Semiarid Ecosystems in Selected Villages of Ramnad District in Tamil Nadu. In N. Ghosh, P. Mukhopadhyay, A. Shah, & M. Panda (Eds.), *Nature, Economy and Society: Understanding the Linkages* (pp. 347–357). Springer, India. https://doi.org/10.1007/978-81-322-2404-4_18

Chapuis, J. L., Boussès, P., & Barnaud, G. (1994). Alien mammals, impact and management in the French subantarctic islands. *Biological Conservation*, *67*(2), 97–104. https://doi.org/10.1016/0006-3207(94)90353-0

Chapuis, J. L., Frenot, Y., & Lebouvier, M. (2004). Recovery of native plant communities after eradication of rabbits from the subantarctic Kerguelen Islands, and influence of climate change. *Biological Conservation*, *117*(2), 167–179. https://doi.org/10.1016/S0006-3207(03)00290-8

Charles, H., & Dukes, J. S. (2007). Impacts of Invasive Species on Ecosystem Services. In W. Nentwig (Ed.), *Biological Invasions* (Vol. 193, pp. 217–237). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-540-36920-2_13

Chikoye, D., Manyong, V. M., & Ekeleme, F. (2000). Characteristics of speargrass (*Imperata cylindrica*) dominated fields in West Africa: Crops, soil properties, farmer perceptions and management strategies. *Crop Protection*, *19*(7), 481-487. https://doi.org/10.1016/S0261-2194(00)00044-2

Chikuni, M. F., Dudley, C. O., & Sambo, E. Y. (2004). *Prosopis glandulosa* Torrey (Leguminosae-Mimosoidae) at Swang'oma, Lake Chilwa plain: A blessing in disguise? *Malawi Journal of Science and Technology*, *7*(1), 10–16. https://www.ajol.info/index.php/mjst/article/view/106762

Chimweta, M., Nyakudya, I. W., Jimu, L., & Mashingaidze, A. B. (2020). Fall armyworm [*Spodoptera frugiperda* (J.E. Smith)] damage in maize: Management options for flood-recession cropping smallholder farmers. *International Journal of Pest Management*, *66*(2), 142–154. https://doi.org/10.1080/09670874.2019.1577514

Chinchio, E., Crotta, M., Romeo, C., Drewe, J. A., Guitian, J., & Ferrari, N. (2020). Invasive alien species and disease risk: An open challenge in public and animal health. *PLoS Pathogens*, *16*(10), e1008922. https://doi.org/10.1371/journal.ppat.1008922

Chown, S. L., & Block, W. (1997). Comparative nutritional ecology of grass-feeding in a sub-Antarctic beetle: The impact of introduced species on *Hydromedion sparsutum* from South Georgia. *Oecologia*, *111*(2), 216–224. https://doi.org/10.1007/s004420050228

Christensen, C. C., Hayes, K. A., & Yeung, N. W. (2021). Taxonomy, Conservation, and the Future of Native Aquatic Snails in the Hawaiian Islands. *Diversity*, *13*(5), 215. https://doi.org/10.3390/d13050215

Chun, C. L., Ochsner, U., Byappanahalli, M. N., Whitman, R. L., Tepp, W. H., Lin, G., Johnson, E. A., Peller, J., & Sadowsky, M. J. (2013). Association of toxin-producing *Clostridium botulinum* with the macroalga *Cladophora* in the Great Lakes. *Environmental Science & Technology*, *47*(6), 2587–2594. https://doi.org/10.1021/es304743m

Cid-Aguayo, B., Ramirez, A., Sepúlveda, M., & Gomez-Uchida, D. (2021). Invasive Chinook Salmon in Chile: Stakeholder Perceptions and Management Conflicts around a New Common-use Resource. *Environmental Management*, *68*(6), 814–823. https://doi.org/10.1007/s00267-021-01528-0

Ciruna, K. A., Meyerson, L. A., & Gutierrez, A. (2004). *The ecological and socio-economic impacts of invasive alien species in inland water ecosystems.* (Report to the Conservation on Biological Diversity on behalf of the Global Invasive Species Programme; p. 34). Washington DC. https://www.cbd.int/doc/ref/alien/ias-inland-waters-en.pdf

Cóbar-Carranza, A. J., García, R. A., Pauchard, A., & Peña, E. (2014). Effect of *Pinus contorta* invasion on forest fuel properties and its potential implications on the fire regime of *Araucaria araucana* and *Nothofagus antarctica* forests. *Biological Invasions*, *16*(11), 2273–2291. https://doi.org/10.1007/s10530-014-0663-8

Cohen, A. N., & Carlton, J. T. (1995). *Biological Study Nonindigenous Aquatic Species in a United States Estuary: A Case Study of the Biological Invasions of the San Francisco Bay and Delta* (SFEI Contribution No. 185). United States Fish and Wildlife Service.

Cohen, J. M., Civitello, D. J., Venesky, M. D., McMahon, T. A., & Rohr, J. R. (2019). An interaction between climate change and infectious disease drove widespread amphibian declines. *Global Change Biology*, *25*(3), 927–937. https://doi.org/10.1111/gcb.14489

Colautti, R. I., Bailey, S. A., van Overdijk, C. D. A., Amundsen, K., & MacIsaac, H. J. (2006). Characterised and Projected Costs of Nonindigenous Species in Canada. *Biological Invasions*, *8*(1), 45–59. https://doi.org/10.1007/s10530-005-0236-y

Coleman, R. A., & Hockey, P. A. R. (2008). Effects of an alien invertebrate species and wave action on prey selection by African black oystercatchers (*Haematopus moquini*). *Austral Ecology*, *33*(2), 232–240. https://doi.org/10.1111/j.1442-9993.2008.01864.x

Colledge, S., Conolly, J., Dobney, K., Manning, K., & Shennan, S. (Eds.). (2013). *Origins and spread of domestic animals in southwest Asia and Europe* (1st ed., Vol. 59). Routledge. https://doi.org/10.4324/9781315417653

Collin, Y. R. H. (2017). *The relationship between the indigenous peoples of the Americas and the horse: Deconstructing a Eurocentric myth* [Dissertation (Ph.D.), University of Alaska Fairbanks]. https://scholarworks.alaska.edu/handle/11122/7592

Concepcion, G. T., Kahng, S. E., Crepeau, M. W., Franklin, E. C., Coles, S. L., & Toonen, R. J. (2010). Resolving natural ranges and marine invasions in a globally distributed octocoral (genus *Carijoa*). *Marine Ecology Progress Series*, *401*, 113–127. https://doi.org/10.3354/meps08364

Cooper, J., Marais, A. v. N., Bloomer, J. P., & Bester, M. N. (1995). A success story: Breeding of burrowing petrels (Procellariidae) before and after the eradication of feral cats *Felis catus* at Subantarctic Marion Island. *Marine Ornithology*, *23*, 33-37. https://www.semanticscholar.org/paper/A-SUCCESS-STORY%3A-BREEDING-OF-BURROWING-PETRELS-AND-Cooper-Marais/dbc7c9bc0add8004e014c91ebd5845ac9ae00a3f

Coote, T., & Loève, É. (2003). From 61 species to five: Endemic tree snails of the Society Islands fall prey to an ill-judged biological control programme. *Oryx*, *37*(1), 91–96. https://doi.org/10.1017/S0030605303000176

Corrales, X., Coll, M., Ofir, E., Heymans, J. J., Steenbeek, J., Goren, M., Edelist, D., & Gal, G. (2018). Future scenarios of marine resources and ecosystem conditions in the Eastern Mediterranean under the impacts of fishing, alien species and sea warming. *Scientific Reports*, *8*(1), 14284. https://doi.org/10.1038/s41598-018-32666-x

Cortina, C., Aslan, C. E., & Litson, S. J. (2019). Importance of non-native honeybees (*Apis mellifera*) as flower visitors to the Hawaiian tree ʻŌhiʻa lehua (*Metrosideros polymorpha*) across an elevation gradient. *Pacific Science*, *73*(3), 345–355. https://doi.org/10.2984/73.3.3

COSEWIC. (2018). *COSEWIC Assessment and Status Report on the Black Ash Fraxinus* nigra *in Canada.* (p. xii + 95pp). Committee on the Status of Endangered Wildlife in Canada. http://www.registrelep-sararegistry.gc.ca/default.asp?lang=en&n=24F7211B-1

Costanza, K. K. L., Livingston, W. H., Kashian, D. M., Slesak, R. A., Tardif, J. C., Dech, J. P., Diamond, A. K., Daigle, J. J., Ranco, D. J., Neptune, J. S., Benedict, L., Fraver, S. R., Reinikainen, M., & Siegert, N. W. (2017). The Precarious State of a Cultural Keystone Species: Tribal and Biological Assessments of the Role and Future of Black Ash. *Journal of Forestry*, *115*(5), 435–446. https://doi.org/10.5849/jof.2016-034R1

Cowie, R. H., Dillon, R. T., Robinson, D. G., & Smith, J. W. (2009). Alien Non-Marine Snails and Slugs of Priority Quarantine Importance in the United States: A Preliminary Risk Assessment. *American Malacological Bulletin*, *27*(1–2), 113–132. https://doi.org/10.4003/006.027.0210

Cox, G. W. (1999). *Alien Species in North America and Hawaii*. Island Press. https://islandpress.org/books/alien-species-north-america-and-hawaii

Cox, G. W. (2004). *Alien species and evolution: The evolutionary ecology of exotic plants, animals, microbes, and interacting native species*. Island Press. https://books.google.co.jp/books/about/Alien_Species_and_Evolution.html?id=itkbzboah2QC&redir_esc=y

Cox, J. G., & Lima, S. L. (2006). Naiveté and an aquatic-terrestrial dichotomy in the effects of introduced predators. *Trends in Ecology and Evolution*, *21*(12), P674-680. https://doi.org/10.1016/j.tree.2006.07.011

Cox, P. A. (1983). Extinction of the Hawaiian Avifauna Resulted in a Change of Pollinators for the ieie, *Freycinetia arborea*. *Oikos*, *41*(2), 195–199. https://doi.org/10.2307/3544263

Cox, P. A., & Elmqvist, T. (2000). Pollinator Extinction in the Pacific Islands. *Conservation Biology*, *14*(5), 1237–1239. https://www.jstor.org/stable/2641771

Crabtree, P. J. (2016). Zooarchaeology in Oceania: An overview. *Archaeology in Oceania*, *51*(1), 1–6. https://doi.org/10.1002/arco.5089

Crego-Prieto, V., Ardura, A., Juanes, F., Roca, A., Taylor, J. S., & Garcia-Vazquez, E. (2015). Aquaculture and the spread of introduced mussel genes in British Columbia. *Biological Invasions*, *17*(7), 2011–2026. https://doi.org/10.1007/s10530-015-0853-z

Crooks, J. A., Chang, A. L., & Ruiz, G. M. (2011). Aquatic pollution increases the relative success of invasive species. *Biological Invasions*, *13*(1), 165–176. https://doi.org/10.1007/s10530-010-9799-3

Crowley, S. L. (2014). Camels Out of Place and Time: The Dromedary (*Camelus dromedarius*) in Australia. *Anthrozoös*, *27*(2), 191–203. https://doi.org/10.2752/175303714X13903827487449

Crystal-Ornelas, R., Hudgins, E. J., Cuthbert, R. N., Haubrock, P. J., Fantle-Lepczyk, J., Angulo, E., Kramer, A. M., Ballesteros-Mejia, L., Leroy, B., Leung, B., López-López, E., Diagne, C., & Courchamp, F. (2021). Economic costs of biological invasions within North America. *NeoBiota*, *67*, 485–510. https://doi.org/10.3897/neobiota.67.58038

Cucherousset, J., Aymes, J. C., Poulet, N., Santoul, F., & Céréghino, R. (2008). Do native brown trout and non-native brook trout interact reproductively? *Naturwissenschaften*, *95*(7), 647–654. https://doi.org/10.1007/s00114-008-0370-3

Cucherousset, J., Aymes, J. C., Santoul, F., & Céréghino, R. (2007). Stable isotope evidence of trophic interactions between introduced brook trout *Salvelinus fontinalis* and native brown trout *Salmo trutta* in a mountain stream of south-west France. *Journal of Fish Biology*, *71*, 210–223. https://doi.org/10.1111/j.1095-8649.2007.01675.x

Cucherousset, J., & Olden, J. D. (2011). Ecological Impacts of Nonnative Freshwater Fishes. *Fisheries*, *36*(5), Article 5. https://doi.org/10.1080/03632415.2011.574578

Curran, G., Carew, M., & Martin, B. N. (2019). Representations of Indigenous Cultural Property in Collaborative Publishing Projects: The Warlpiri Women's *Yawulyu* Songbooks. *Journal of Intercultural Studies*, *40*(1), 68–84. https://doi.org/10.1080/07256868.2018.1552572

Cuthbert, R. N., Bartlett, A. C., Turbelin, A. J., Haubrock, P. J., Diagne, C., Pattison, Z., Courchamp, F., & Catford, J. A. (2021). Economic costs of biological invasions in the United Kingdom. *NeoBiota*, *67*, 299–328. https://doi.org/10.3897/neobiota.67.59743

Cuthbert, R. N., Diagne, C., Hudgins, E. J., Turbelin, A., Ahmed, D. A., Albert, C., Bodey, T. W., Briski, E., Essl, F., Haubrock, P. J., Gozlan, R. E., Kirichenko, N., Kourantidou, M., Kramer, A. M., & Courchamp, F. (2022). Biological invasion costs reveal insufficient proactive management worldwide. *Science of The Total Environment*, *819*, 153404. https://doi.org/10.1016/j.scitotenv.2022.153404

Cuthbert, R. N., Pattison, Z., Taylor, N. G., Verbrugge, L., Diagne, C., Ahmed, D. A., Leroy, B., Angulo, E., Briski, E., & Capinha, C. (2021). Global economic costs of aquatic invasive alien species. *Science of the Total Environment*, *775*, 145238. https://doi.org/10.1016/j.scitotenv.2021.145238

Czech, B., & Krausman, P. R. (1999). Research notes public opinion on endangered species conservation and policy. *Society & Natural Resources*, *12*(5), 469–479. https://doi.org/10.1080/089419299279542

Dalal, A., Cuthbert, R. N., Dick, J. T. A., Sentis, A., Laverty, C., Barrios-O'Neill, D., Perea, N. O., Callaghan, A., & Gupta, S. (2020). Prey size and predator density modify impacts by natural enemies towards mosquitoes. *Ecological Entomology*, *45*(3), 423–433. https://doi.org/10.1111/een.12807

Daly, J. W., Myers, C. W., & Whittaker, N. (1987). Further classification of skin alkaloids from neotropical poison frogs (dendrobatidae), with a general survey of toxic/noxious substances in the amphibia. *Toxicon: Official Journal of the International Society on Toxinology*, *25*(10), 1023–1095. https://doi.org/10.1016/0041-0101(87)90265-0

Damasceno, G. A. de B., Ferrari, M., & Giordani, R. B. (2017). *Prosopis juliflora* (SW) D.C., an invasive specie at the Brazilian Caatinga: Phytochemical, pharmacological, toxicological and technological overview. *Phytochemistry Reviews*, *16*(2), 309–331. https://doi.org/10.1007/s11101-016-9476-y

Dana, E. D., Jeschke, J. M., & García de Lomas, J. (2014). Decision tools for managing biological invasions: Existing biases and future needs. *ORYX*, *48*(1), 56–63. https://doi.org/10.1017/S0030605312001263

D'Antonio, C. M., & Dudley, T. L. (1995). Biological Invasions as Agents of Change on Islands *Versus* Mainlands. In P. M. Vitousek, L. L. Loope, & H. Adsersen (Eds.), *Islands: Biological Diversity and Ecosystem Function* (pp. 103–121). Springer. https://doi.org/10.1007/978-3-642-78963-2_9

Darwall, W., Bremerich, V., De Wever, A., Dell, A. I., Freyhof, J., Gessner, M. O., Grossart, H.-P., Harrison, I., Irvine, K., Jähnig, S. C., Jeschke, J. M., Lee, J. J., Lu, C., Lewandowska, A. M., Monaghan, M. T., Nejstgaard, J. C., Patricio, H., Schmidt-Kloiber, A., Stuart, S. N., … Weyl, O. (2018). The *Alliance for Freshwater Life*: A global call to unite efforts for freshwater biodiversity science and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *28*(4), 1015–1022. https://doi.org/10.1002/aqc.2958

Darwin, C. (1839). *The Voyage of the Beagle*. http://test.darwin-online.org.uk/converted/published/1838-1843_Zoology_F9/1842_Zoology_F9.4.html

Dassonville, N., Guillaumaud, N., Piola, F., Meerts, P., & Poly, F. (2011). Niche construction by the invasive Asian knotweeds (species complex *Fallopia*): Impact on activity, abundance and community structure of denitrifiers and nitrifiers. *Biological Invasions*, *13*(5), 1115–1133. https://doi.org/10.1007/s10530-011-9954-5

Davidson, A. D., Campbell, M. L., Hewitt, C. L., & Schaffelke, B. (2015). Assessing the impacts of nonindigenous marine macroalgae: An update of current knowledge. *Botanica Marina*, *58*(2), 55–79. https://doi.org/10.1515/bot-2014-0079

Davies, H. F., McCarthy, M. A., Firth, R. S. C., Woinarski, J. C. Z., Gillespie, G. R., Andersen, A. N., Geyle, H. M., Nicholson, E., & Murphy, B. P. (2017). Top-down control of species distributions: Feral cats driving the regional extinction of a threatened rodent in northern Australia. *Diversity and Distributions*, *23*(3), 272–283. https://doi.org/10.1111/ddi.12522

Davis, E. S., Kelly, R., Maggs, C. A., & Stout, J. C. (2018). Contrasting impacts of highly invasive plant species on flower-visiting insect communities. *Biodiversity and Conservation*, *27*(8), 2069–2085. https://doi.org/10.1007/s10531-018-1525-y

Dawson, W., Peyton, J. M., Pescott, O. L., Adriaens, T., Cottier-Cook, E. J., Frohlich, D. S., Key, G., Malumphy, C., Martinou, A. F., Minchin, D., Moore, N., Rabitsch, W., Rorke, S. L., Tricarico, E., Turvey, K. M. A., Winfield, I. J., Barnes, D. K. A., Baum, D., Bensusan, K., … Roy, H. E. (2022). Horizon scanning for potential invasive non-native species across the United Kingdom Overseas Territories. *Conservation Letters*. https://doi.org/10.1111/conl.12928

Day, R., Abrahams, P., Bateman, M., Beale, T., Clottey, V., Cock, M., Colmenarez, Y., Corniani, N., Early, R., Godwin, J., Gomez, J., Moreno, P. G., Murphy, S. T., Oppong-Mensah, B., Phiri, N., Pratt, C., Silvestri, S., & Witt, A. (2017). Fall Armyworm: Impacts and Implications for Africa. *Outlooks on Pest Management*, *28*(5), 196–201. https://doi.org/10.1564/v28_oct_02

Dayal, V. (2007). Social diversity and ecological complexity: How an invasive tree could affect diverse agents in the land of the tiger. *Environment and Development Economics*, *12*(4), 553–571. https://doi.org/10.1017/S1355770X07003695

de Caralt, S., & Cebrian, E. (2013). Impact of an invasive alga (*Womersleyella setacea*) on sponge assemblages: Compromising the viability of future populations. *Biological Invasions*, *15*(7), 1591–1600. https://doi.org/10.1007/s10530-012-0394-7

de Greef, K., Griffiths, C. L., & Zeeman, Z. (2013). Deja vu? A second mytilid mussel, *Semimytilus algosus*, invades South Africa's west coast. *African Journal of Marine Science*, *35*(3), 307–313. https://doi.org/10.2989/1814232X.2013.829789

De Groote, H., Kimenju, S. C., Munyua, B., Palmas, S., Kassie, M., & Bruce, A. (2020). Spread and impact of fall armyworm (*Spodoptera frugiperda* J.E. Smith) in maize production areas of Kenya. *Agriculture, Ecosystems and Environment*, *292*. https://doi.org/10.1016/j.agee.2019.106804

de Montaudouin, X., & Sauriau, P.-G. (1999). The proliferating Gastropoda *Crepidula fornicata* may stimulate macrozoobenthic diversity. *Journal of the Marine Biological Association of the UK*, *79*(6), 1069–1077. https://hal.archives-ouvertes.fr/hal-01854016

de Rivera, C. E., Grosholz, E. D., & Ruiz, G. M. (2011). Multiple and long-term effects of an introduced predatory crab. *Marine Ecology Progress Series*, *429*, 145–155. https://doi.org/10.3354/meps09101

De Silva, S. S. (2012). Aquaculture: A newly emergent food production sector—and perspectives of its impacts on biodiversity and conservation. *Biodiversity and Conservation*, *21*(12), 3187–3220. https://doi.org/10.1007/s10531-012-0360-9

De Silva, S. S., Nguyen, T. T. T., Turchini, G. M., Amarasinghe, U. S., & Abery, N. W. (2009). Alien Species in Aquaculture and Biodiversity: A Paradox in Food Production. *AMBIO: A Journal of the Human Environment*, *38*(1), 24–28. https://doi.org/10.1579/0044-7447-38.1.24

de Souza, T. A. F., de Andrade, L. A., Freitas, H., & da Silva Sandim, A. (2018). Biological invasion influences the outcome of plant-soil feedback in the invasive plant species from the brazilian semi-arid. *Microbial Ecology*, *76*, 102–112. https://doi.org/10.1007/s00248-017-0999-6

de Villèle, X., & Verlaque, M. (1995). Changes and Degradation in a *Posidonia oceanica* Bed Invaded by the Introduced Tropical Alga *Caulerpa taxifolia* in the North Western Mediterranean. *Botanica Marina*, *38*. https://www.degruyter.com/document/doi/10.1515/botm.1995.38.1-6.79/html

de Wit, M. P., Crookes, D. J., & van Wilgen, B. W. (2001). Conflicts of interest in environmental management: Estimating the costs and benefits of a tree invasion. *Biological Invasions*, *3*(2), 12. https://link.springer.com/article/10.1023/A:1014563702261

Déchamp, C. (1999). Ragweed, a biological pollutant: Current and desirable legal implications in France and Europe. *Revue Française d'Allergologie et d'Immunologie Clinique*, *39*(4), 289–294. https://doi.org/10.1016/S0335-7457(99)80056-2

Dehnen-Schmutz, K., Pescott, O. L., Booy, O., & Walker, K. J. (2022). Integrating expert knowledge at regional and national scales improves impact assessments of non-native species. *NeoBiota*, *77*, 79–100. https://doi.org/10.3897/neobiota.77.89448

DellaValle, C. T., Triche, E. W., Leaderer, B. P., & Bell, M. L. (2012). Effects of ambient pollen concentrations on frequency and severity of asthma symptoms among asthmatic children. *Epidemiology*, *23*(1), 55–63. https://doi.org/10.1097/EDE.0b013e31823b66b8

Delnatte, C., & Meyer, J.-Y. (2012). Plant introduction, naturalization, and invasion in French Guiana (South America). *Biological Invasions*, *14*(5), 915–927. https://doi.org/10.1007/s10530-011-0129-1

DeRoy, E., Scott, R., Hussey, N., & MacIsaac, H. (2020). High predatory efficiency and abundance drive expected ecological impacts of a marine invasive fish. *Marine Ecology Progress Series*, *637*, 195–208. https://doi.org/10.3354/meps13251

deShazo, R. D., & Banks, W. A. (1994). Medical consequences of multiple fire ant stings occurring indoors. *The Journal of Allergy and Clinical Immunology*, *93*(5), 847–850. https://doi.org/10.1016/0091-6749(94)90376-x

deShazo, R. D., Butcher, B. T., & Banks, W. A. (1990). Reactions to the Stings of the Imported Fire Ant. *New England Journal of Medicine*, *323*(7), 462–466. https://doi.org/10.1056/NEJM199008163230707

deShazo, R. D., Williams, D. F., & Moak, E. S. (1999). Fire ant attacks on residents in health care facilities: A report of two cases. *Annals of Internal Medicine*, *131*(6), 424–429. https://doi.org/10.7326/0003-4819-131-6-199909210-00005

Desroy, N., Dubois, S. F., Fournier, J., Ricquiers, L., Mao, P. L., Guerin, L., Gerla, D., Rougerie, M., & Legendre, A. (2011). The conservation status of *Sabellaria alveolata* (L.) (*Polychaeta: Sabellariidae*) reefs in the Bay of Mont-Saint-Michel. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *21*(5), 462–471. https://doi.org/10.1002/aqc.1206

Dexter, N., & Murray, A. (2009). The impact of fox control on the relative abundance of forest mammals in East Gippsland, Victoria. *Wildlife Research*, *36*(3), 252–261. https://doi.org/10.1071/WR08135

Diagne, C., Catford, J. A., Essl, F., Nuñez, M. A., & Courchamp, F. (2020). What are the economic costs of biological invasions? A complex topic requiring international and interdisciplinary expertise. *NeoBiota*, *63*, 25–37. https://doi.org/10.3897/neobiota.63.55260

Diagne, C., Leroy, B., Gozlan, R. E., Vaissière, A.-C., Assailly, C., Nuninger, L., Roiz, D., Jourdain, F., Jarić, I., & Courchamp, F. (2020). InvaCost, a public database of the economic costs of biological invasions worldwide. *Scientific Data*, *7*(1), 277. https://doi.org/10.1038/s41597-020-00586-z

Diagne, C., Leroy, B., Vaissière, A.-C., Gozlan, R. E., Roiz, D., Jarić, I., Salles, J.-M., Bradshaw, C. J. A., & Courchamp, F. (2021). High and rising economic costs of biological invasions worldwide. *Nature*, *592*(7855), 571–576. https://doi.org/10.1038/s41586-021-03405-6

Diagne, C., Turbelin, A. J., Moodley, D., Novoa, A., Leroy, B., Angulo, E., Adamjy, T., Dia, C. A. K. M., Taheri, A., Tambo, J., Dobigny, G., & Courchamp, F. (2021). The economic costs of biological invasions in Africa: A growing but neglected threat? *NeoBiota*, *67*, 11–51. https://doi.org/10.3897/neobiota.67.59132

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., Hill, R., Chan, K. M. A., Baste, I. A., Brauman, K. A., Polasky, S., Church, A., Lonsdale, M., Larigauderie, A., Leadley, P. W., van Oudenhoven, A. P. E., van der Plaat, F., Schröter, M., Lavorel, S., … Shirayama, Y. (2018). An inclusive approach to assesse nature's contributions to people. *Science*, *359*(6373), 270–272. https://doi.org/10.1126/science.aap8826

Dick, J. T. A., & Platvoet, D. (2000). Invading predatory crustacean *Dikerogammarus villosus* eliminates both native and exotic species. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, *267*(1447), 977–983. https://doi.org/10.1098/rspb.2000.1099

Dijkstra, J., & Nolan, R. (2011). Potential of the invasive colonial ascidian, *Didemnum vexillum*, to limit escape response of the sea scallop, *Placopecten magellanicus*. *Aquatic Invasions*, *6*(4), 451–456. https://doi.org/10.3391/ai.2011.6.4.10

Dilley, B. J., Schoombie, S., Schoombie, J., & Ryan, P. G. (2016). 'Scalping' of albatross fledglings by introduced mice spreads rapidly at Marion Island. *Antarctic Science*, *28*(2), 73–80. https://doi.org/10.1017/S0954102015000486

Dilley, B. J., Schoombie, S., Stevens, K., Davies, D., Perold, V., Osborne, A., Schoombie, J., Brink, C. W., Carpenter-Kling, T., & Ryan, P. G. (2018). Mouse predation affects breeding success of burrow-nesting petrels at sub-Antarctic Marion Island. *Antarctic Science*, *30*(2), 93–104. https://doi.org/10.1017/S0954102017000487

Dilley, B. J., Schramm, M., & Ryan, P. G. (2017). Modest increases in densities of burrow-nesting petrels following the removal of cats *(Felis catus)* from Marion Island. *Polar Biology*, *40*(3), 625–637. https://doi.org/10.1007/s00300-016-1985-z

Dinasquet, J., Granhag, L. M., & Riemann, L. (2012). Stimulated bacterioplankton growth and selection for certain bacterial taxa in the vicinity of the ctenophore *Mnemiopsis leidyi*. *Frontiers in Microbiology*, *3*, 302. https://doi.org/10.3389/fmicb.2012.00302

Dixit, S., & Dhote, S. (2010). Evaluation of uptake rate of heavy metals by *Eichhornia crassipes* and *Hydrilla verticillata*. *Environmental Monitoring and Assessment*, *169*(1–4), 367–374. https://doi.org/10.1007/s10661-009-1179-z

Dobhal, P. K., Kohli, R. K., & Batish, D. R. (2010). Evaluation of the impact of *Lantana camara* L. invasion, on four major woody shrubs, along Nayar river of Pauri Garhwal, in Uttarakhand Himalaya. *International Journal of Biodiversity and Conservation*, *2*(7), 155–161. https://www.semanticscholar.org/paper/Evaluation-of-the-impact-of-Lantana-camara-L.-on-of-Dobhal-Kohli/00a99959ad23767f877e976b314d218db72c0ecb

Doherty, T. S., Dickman, C. R., Johnson, C. N., Legge, S. M., Ritchie, E. G., & Woinarski, J. C. Z. (2017). Impacts and management of feral cats *Felis catus* in Australia. *Mammal Review*, *47*(2), 83–97. https://doi.org/10.1111/mam.12080

Doherty, T. S., Dickman, C. R., Nimmo, D. G., & Ritchie, E. G. (2015). Multiple threats, or multiplying the threats? Interactions between invasive predators and other ecological disturbances. *Biological Conservation*, *190*, 60–68. https://doi.org/10.1016/j.biocon.2015.05.013

Doherty, T. S., Glen, A. S., Nimmo, D. G., Ritchie, E. G., & Dickman, C. R. (2016). Invasive predators and global biodiversity loss. *Proceedings of the National Academy of Sciences*, *113*(40), 11261–11265. https://doi.org/10.1073/pnas.1602480113

Doody, J. S., Castellano, C. M., Rhind, D., & Green, B. (2013). Indirect facilitation of a native mesopredator by an invasive species: Are cane toads re-shaping tropical riparian communities? *Biological Invasions*, *15*(3), 559–568. https://doi.org/10.1007/s10530-012-0308-8

dos Santos, L. L., do Nascimento, A. L. B., Vieira, F. J., da Silva, V. A., Voeks, R., & Albuquerque, U. P. (2014). The Cultural Value of Invasive Species: A Case Study from Semi–Arid Northeastern Brazil. *Economic Botany*, *68*(3), 283–300. https://doi.org/10.1007/s12231-014-9281-8

Dove, C. J., Snow, R. W., Rochford, M. R., & Mazzotti, F. J. (2011). Birds Consumed by the Invasive Burmese Python (*Python molurus bivittatus*) in Everglades National Park, Florida, USA. *The Wilson Journal of Ornithology*, *123*(1), 126–131. https://doi.org/10.1676/10-092.1

Drake, D. R., & Hunt, T. L. (2009). Invasive rodents on islands: Integrating historical and contemporary ecology. *Biological Invasions*, *11*, 1483–1487. https://doi.org/10.1007/s10530-008-9392-1

Driscoll, D. A., Catford, J. A., Barney, J. N., Hulme, P. E., Inderjit, Martin, T. G., Pauchard, A., Pyšek, P., Richardson, D. M., Riley, S., & Visser, V. (2014). New pasture plants intensify invasive species risk. *Proceedings of the National Academy of Sciences*, *111*(46), 16622–16627. https://doi.org/10.1073/pnas.1409347111

Duboscq-Carra, V. G., Fernandez, R. D., Haubrock, P. J., Dimarco, R. D., Angulo, E., Ballesteros-Mejia, L., Diagne, C., Courchamp, F., & Nuñez, M. A. (2021). Economic impact of invasive alien species in Argentina: A first national synthesis. *NeoBiota*, *67*, 329–348. https://doi.org/10.3897/neobiota.67.63208

Dueñas, M.-A., Hemming, D. J., Roberts, A., & Diaz-Soltero, H. (2021). The threat of invasive species to IUCN-listed critically endangered species: A systematic review. *Global Ecology and Conservation*, *26*, e01476. https://doi.org/10.1016/j.gecco.2021.e01476

Duenn, P., Salpeteur, M., & Reyes-García, V. (2017). Rabari Shepherds and the Mad Tree: The Dynamics of Local Ecological Knowledge in the Context of *Prosopis juliflora* Invasion in Gujarat, India. *Journal of Ethnobiology*, *37*(3), 561–580. https://doi.org/10.2993/0278-0771-37.3.561

Duffy, D. C., & Capece, P. (2012). Biology and Impacts of Pacific Island Invasive Species. 7. The Domestic Cat (*Felis catus*). *Pacific Science*, *66*(2), 173–212. https://doi.org/10.2984/66.2.7

Duffy, M. R., Chen, T.-H., Hancock, W. T., Powers, A. M., Kool, J. L., Lanciotti, R. S., Pretrick, M., Marfel, M., Holzbauer, S., Dubray, C., Guillaumot, L., Griggs, A., Bel, M., Lambert, A. J., Laven, J., Kosoy, O., Panella, A., Biggerstaff, B. J., Fischer, M., & Hayes, E. B. (2009). Zika virus outbreak on Yap Island, Federated States of Micronesia. *The New England Journal of Medicine*, *360*(24), 2536–2543. https://doi.org/10.1056/NEJMoa0805715

Dumay, O., Fernandez, C., & Pergent, G. (2002). Primary production and vegetative cycle in *Posidonia oceanica* when in competition with the green algae *Caulerpa taxifolia* and *Caulerpa racemosa*. *Journal of the Marine Biological Association of the United Kingdom*, *82*(3), 379–387. https://www.cambridge.org/core/journals/journal-of-the-marine-biological-association-of-the-united-kingdom/article/abs/primary-production-and-vegetative-cycle-in-posidonia-oceanica-when-in-competition-with-the-green-algae-*Caulerpa*-taxifolia-and-*Caulerpa*-racemosa-/79FD06427C9849898A17AF6422B19A49

Duncan, R. P., & Blackburn, T. M. (2007). Causes of extinction in island birds. *Animal Conservation*, *10*(2), 149–150. https://doi.org/10.1111/j.1469-1795.2007.00110.x

Dzikiti, S., Schachtschneider, K., Naiken, V., Gush, M., Moses, G., & Le Maitre, D. C. (2013). Water relations and the effects of clearing invasive *Prosopis* trees on groundwater in an arid environment in the Northern Cape, South Africa. *Journal of Arid Environments*, *90*, 103–113. https://doi.org/10.1016/j.jaridenv.2012.10.015

Effler, P. V., Pang, L., Kitsutani, P., Vorndam, V., Nakata, M., Ayers, T., Elm, J., Tom, T., Reiter, P., Rigau-Perez, J. G., Hayes, J. M., Mills, K., Napier, M., Clark, G. G., Gubler, D. J., & Hawaii Dengue Outbreak Investigation Team. (2005). Dengue fever, Hawaii, 2001-2002. *Emerging Infectious Diseases*, *11*(5), 742–749. https://doi.org/10.3201/eid1105.041063

EFSA PLH Panel (EFSA Panel on Plant Health). (2014). Scientific Opinion on the pest categorisation of *Cryphonectria parasitica* (Murrill) Barr. *EFSA Journal*, *2014;12(10):3859*. https://doi.org/10.2903/j.efsa.2014.3859

Eggertsen, M., Tano, S. A., Chacin, D. H., Eklöf, J. S., Larsson, J., Berkström, C., Buriyo, A. S., & Halling, C. (2021). Different environmental variables predict distribution and cover of the introduced red seaweed *Eucheuma denticulatum* in two geographical locations. *Biological Invasions*, *23*(4), 1049–1067. https://doi.org/10.1007/s10530-020-02417-z

Eklöf, J. S., de la Torre Castro, M., Adelsköld, L., Jiddawi, N. S., & Kautsky, N. (2005). Differences in macrofaunal and seagrass assemblages in seagrass beds with and without seaweed farms. *Estuarine, Coastal and Shelf Science*, *63*(3), 385–396. https://doi.org/10.1016/j.ecss.2004.11.014

Eklöf, J. S., de la Torre-Castro, M., Nilsson, C., & Rönnbäck, P. (2006). How do seaweed farms influence local fishery catches in a seagrass-dominated setting in Chwaka Bay, Zanzibar? *Aquatic Living Resources*, *19*(2), 137–147. https://doi.org/10.1051/alr:2006013

Eldridge, D. J., & James, A. I. (2009). Soil-disturbance by native animals plays a critical role in maintaining healthy Australian landscapes. *Ecological Management & Restoration*, *10*(s1), S27–S34. https://doi.org/10.1111/j.1442-8903.2009.00452.x

Ellender, B. R., & Weyl, O. L. F. (2014). A review of current knowledge, risk and ecological impacts associated with non-native freshwater fish introductions in South Africa. *Aquatic Invasions*, *9*(2), 117–132. https://doi.org/10.3391/ai.2014.9.2.01

Ellender, B. R., Woodford, D. J., Weyl, O. L. F., & Cowx, I. G. (2014). Managing conflicts arising from fisheries enhancements based on non-native fishes in southern Africa. *Journal of Fish Biology*, *85*(6), 1890–1906. https://doi.org/10.1111/jfb.12512

Elliott, P., Aldridge, David. C., Moggridge, G. D., & Chipps, M. (2005). The increasing efffects of zebra mussels on water installations in England. *Water and Environment Journal*, *19*(4), 367–375. https://doi.org/10.1111/j.1747-6593.2005.tb00575.x

Elton, C. S. (1958). *The Ecology of Invasions by Animals and Plants*. University of Chicago Press. http://link.springer.com/10.1007/978-1-4899-7214-9

Emberlin, J. (1994). The effects of patterns in climate and pollen abundance on allergy. *Allergy*, *49*(18 Suppl), 15–20. https://doi.org/10.1111/j.1398-9995.1994.tb04233.x

Emery-Butcher, H. E., Beatty, S. J., & Robson, B. J. (2020). The impacts of invasive ecosystem engineers in freshwaters: A review. *Freshwater Biology*, *65*(5), 999–1015. https://doi.org/10.1111/fwb.13479

Englund, R. E. (1999). The Impacts of Introduced Poeciliid Fish and Odonata on the Endemic Megalagrion (Odonata) Damselflies of Oahu Island, Hawaii. *Journal of Insect Conservation*, *3*(3), 225–243. https://doi.org/10.1023/A:1009651922486

Ens, E. J., Fisher, J., & Costello, O. (Eds.). (2015). *Indigenous People and Invasive Species: Perceptions, management, challenges and uses*. IUCN Commission on Ecosystem Management Community Booklet. https://ipm.ifas.ufl.edu/pdfs/ens_et_al_2015_indigenous_people_and_invasive_species_iucn_cem_ecosystems_and_invasiv.pdf

Ens, E. J., Pert, P., Clarke, P. A., Budden, M., Clubb, L., Doran, B., Douras, C., Gaikwad, J., Gott, B., Leonard, S., Locke, J., Packer, J., Turpin, G., & Wason, S. (2015). Indigenous biocultural knowledge in ecosystem science and management: Review and insight from Australia. *Biological Conservation*, *181*, 133–149. https://doi.org/10.1016/j.biocon.2014.11.008

Enserink, M. (2006). Massive Outbreak Draws Fresh Attention to Little-Known Virus. *Science*. https://doi.org/10.1126/SCIENCE.311.5764.1085A

Essian, D. A., Chipault, J. G., Lafrancois, B. M., & Leonard, J. B. (2016). Gut content analysis of Lake Michigan waterbirds in years with avian botulism type E mortality, 2010–2012. *Journal of Great Lakes Research*, *42*(5), 1118–1128. https://doi.org/10.1016/j.jglr.2016.07.027

Essl, F., Dullinger, S., Rabitsch, W., Hulme, P. E., Hülber, K., Jarošík, V., Kleinbauer, I., Krausmann, F., Kühn, I., Nentwig, W., Vilà, M., Genovesi, P., Gherardi, F., Desprez-Loustau, M.-L., Roques, A., & Pyšek, P. (2011). Socioeconomic legacy yields an invasion debt. *Proceedings of the National Academy of Sciences*, *108*(1), 203–207. https://doi.org/10.1073/pnas.1011728108

Essl, F., Lenzner, B., Bacher, S., Bailey, S., Capinha, C., Daehler, C., Dullinger, S., Genovesi, P., Hui, C., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kühn, I., Leung, B., Liebhold, A., Liu, C., MacIsaac, H. J., Meyerson, L. A., Nuñez, M. A., … Roura-Pascual, N. (2020). Drivers of future alien species impacts: An expert-based assessment. *Global Change Biology*, *26*(9), 4880–4893. https://doi.org/10.1111/gcb.15199

Essl, F., Lenzner, B., Courchamp, F., Dullinger, S., Jeschke, J. M., Kühn, I., Leung, B., Moser, D., Roura-Pascual, N., & Seebens, H. (2019). Introducing AlienScenarios: A project to develop scenarios and models of biological invasions for the 21st century. *NeoBiota*, *45*, 1–17. https://doi.org/10.3897/neobiota.45.33366

Essl, F., Mang, T., & Moser, D. (2012). Ancient and recent alien species in temperate forests: Steady state and time lags. *Biological Invasions*, *14*(7), 1331–1342. https://doi.org/10.1007/s10530-011-0156-y

Evans, T., Blackburn, T. M., Jeschke, J. M., Probert, A. F., & Bacher, S. (2020). Application of the Socio-Economic Impact Classification for Alien Taxa (SEICAT) to a global assessment of alien bird impacts. *NeoBiota*, *62*, 123–142. https://doi.org/10.3897/neobiota.62.51150

Evans, T., Kumschick, S., & Blackburn, T. M. (2016). Application of the Environmental Impact Classification for Alien Taxa (EICAT) to a global assessment of alien bird impacts. *Diversity and Distributions*, *22*(9), 919–931. https://doi.org/10.1111/ddi.12464

Ezeamuzie, C. I., Thomson, M. S., Al-Ali, S., Dowaisan, A., Khan, M., & Hijazi, Z. (2000). Asthma in the desert: Spectrum of the sensitizing aeroallergens. *Allergy*, *55*(2), 157–162. https://doi.org/10.1034/j.1398-9995.2000.00375.x

Fabricante, J. R., Araújo, K. C. T. de, Andrade, L. A. de, & Ferreira, J. V. A. (2012). Invasão biológica de *Artocarpus heterophyllus* Lam. (Moraceae) em um fragmento de Mata Atlântica no Nordeste do Brasil: Impactos sobre a fitodiversidade e os solos dos sítios invadidos. *Acta Botanica Brasilica*, *26*(2), 399–407. https://doi.org/10.1590/S0102-33062012000200015

Fache, E. (2021). 5 Mediation between Indigenous and Non-Indigenous Knowledge Systems: Another Analysis of “Two-Way” Conservation in Northern Australia. In *Entangled Territorialities* (pp. 91–116). University of Toronto Press. https://www.degruyter.com/document/doi/10.3138/9781487513764-006/html

Fancourt, B. A., Hawkins, C. E., Cameron, E. Z., Jones, M. E., & Nicol, S. C. (2015). Devil Declines and Catastrophic Cascades: Is Mesopredator Release of Feral Cats Inhibiting Recovery of the Eastern Quoll? *PLoS ONE*, *10*(3), e0119303. https://doi.org/10.1371/journal.pone.0119303

Fantle-Lepczyk, J. E., Haubrock, P. J., Kramer, A. M., Cuthbert, R. N., Turbelin, A. J., Crystal-Ornelas, R., Diagne, C., & Courchamp, F. (2021). Economic costs of biological invasions in the United States. *Science of The Total Environment*, 151318. https://doi.org/10.1016/j.scitotenv.2021.151318

FAO. (2018). *The Republic of Namibia: Fall armyworm impact and needs assessment, 2018* (p. 52). Food and Agriculture Organisation of the United Nations. https://www.fao.org/documents/card/ru/c/I9556EN/

FAO. (2020). *The State of the World Fisheries and Aquaculture 2020. Sustainability in action* (p. 224). Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/ca9229en

Fares, R. C. G., Souza, K. P. R., Añez, G., & Rios, M. (2015). Epidemiological Scenario of Dengue in Brazil. *BioMed Research International*, *2015*, 321873. https://doi.org/10.1155/2015/321873

Faulkes, Z. (2010). The spread of the parthenogenetic marbled crayfish, Marmorkrebs (*Procambarus* sp.), in the North American pet trade. *Aquatic Invasions*, *5*(4), 447–450. https://doi.org/10.3391/ai.2010.5.4.16

Fenetahun, Y., Xu, X. W., & Wang, Y. D. (2020). Analysis of eco-environmental vulnerability: Implication for bush encroachment and livestock population dynamics of the teltele rangeland, southern, Ethiopia. *Applied Ecology and Environmental Research*, *18*(5), 7255–7278. https://doi.org/10.15666/aeer/1805_72557278

Fèvre, E. M., Bronsvoort, B. M. de C., Hamilton, K. A., & Cleaveland, S. (2006). Animal movements and the spread of infectious diseases. *Trends in Microbiology*, *14*(3), 125–131. https://doi.org/10.1016/j.tim.2006.01.004

Ficetola, G. F., Siesa, M. E., Manenti, R., Bottoni, L., De Bernardi, F., & Padoa-Schioppa, E. (2011). Early assessment of the impact of alien species: Differential consequences of an invasive crayfish on adult and larval amphibians. *Diversity and Distributions*, *17*(6), 1141–1151. https://doi.org/10.1111/j.1472-4642.2011.00797.x

Fillios, M., Crowther, M. S., & Letnic, M. (2012). The impact of the dingo on the thylacine in Holocene Australia. *World Archaeology*, *44*(1), 118–134. https://doi.org/10.1080/00438243.2012.646112

Finenko, G. A., Abolmasova, G. I., Datsyk, N. A., & Anninsky, B. E. (2015). The effect of the ctenophore *Mnemiopsis leidyi* A. Agassiz, 1865 (Ctenophora: Lobata) on the population density and species composition of mesoplankton in inshore waters of the Crimea. *Russian Journal of Marine Biology*, *41*(4), 260–271. https://link.springer.com/article/10.1134/S1063074015020042

Finenko, G. A., Abolmasova, G. I., Romanova, Z. A., Datsyk, N. A., & Anninskii, B. E. (2013). Population dynamics of the ctenophore *Mnemiopsis leidyi* and its impact on the zooplankton in the coastal regions of the Black Sea of the Crimean coast in 2004–2008. *Oceanology*, *53*(1), 80–88. https://link.springer.com/article/10.1134/S0001437012050074

Finenko, G. A., Anninsky, B. E., & Datzyk, N. A. (2018). Trophic characteristics of *Mnemiopsis leidyi* and its impact on the plankton Community in Black Sea coastal waters. *Oceanology*, *58*(6), 817–824. https://link.springer.com/article/10.1134/S0001437018060048

Fiori, E., Benzi, M., Ferrari, C. R., & Mazziotti, C. (2019). Zooplankton community structure before and after *Mnemiopsis leidyi* arrival. *Journal of Plankton Research*, *41*(6), 803–820. https://doi.org/10.1093/plankt/fbz060

Fischer, J. R. (2007). Cattle in Hawai'i: Biological and Cultural Exchange. *Pacific Historical Review*, *76*(3), 347–372. https://doi.org/10.1525/phr.2007.76.3.347

Fisher, D. O., Johnson, C. N., Lawes, M. J., Fritz, S. A., McCallum, H., Blomberg, S. P., VanDerWal, J., Abbott, B., Frank, A., Legge, S., Letnic, M., Thomas, C. R., Fisher, A., Gordon, I. J., & Kutt, A. (2014). The current decline of tropical marsupials in Australia: Is history repeating? *Global Ecology and Biogeography*, *23*(2), 181–190. https://doi.org/10.1111/geb.12088

Fisher, M. C., & Garner, T. W. J. (2020). Chytrid fungi and global amphibian declines. *Nature Reviews Microbiology*, *18*(6), 332–343. https://doi.org/10.1038/s41579-020-0335-x

Fletcher, L. M., Forrest, B. M., & Bell, J. J. (2013). Impacts of the invasive ascidian *Didemnum vexillum* on green-lipped mussel *Perna canaliculus* aquaculture in New Zealand. *Aquaculture Environment Interactions*, *4*(1), 17–30. https://doi.org/10.3354/aei00069

Floyd, T., & Williams, J. (2004). Impact of green crab (*Carcinus maenas* L.) predation on a population of soft-shell clams (*Mya arenaria* L.) in the southern Gulf of St. Lawrence. *Journal of Shellfish Research*, *23*(2), 457+. https://link.gale.com/apps/doc/A123080672/AONE?u=anon~cda008c3&sid=googleScholar&xid=d1887550

Fonseca, D. M., Lapointe, D. A., & Fleischer, R. C. (2000). Bottlenecks and multiple introductions: Population genetics of the vector of avian malaria in Hawaii. *Molecular Ecology*, *9*(11), 1803–1814. https://doi.org/10.1046/j.1365-294x.2000.01070.x

Ford, H. A., Barrett, G. W., Saunders, D. A., & Recher, H. F. (2001). Why have birds in the woodlands of southern Australia declined? *Biological Conservation*, *97*(1), 71–88. https://doi.org/10.1016/S0006-3207(00)00101-4

Fordham, D., Georges, A., Corey, B., & Brook, B. W. (2006). Feral pig predation threatens the indigenous harvest and local persistence of snake-necked turtles in northern Australia. *Biological Conservation*, *133*(3), 379–388. https://doi.org/10.1016/j.biocon.2006.07.001

Forest Peoples Programme, International Indigenous Forum on Biodiversity, Indigenous Women's Biodiversity Network, Centres of Distinction on Indigenous and Local Knowledge, & Secretariat of the CBD. (2020). *Local Biodiversity Outlooks 2: The contributions of indigenous peoples and local communities to the implementation of the Strategic Plan for Biodiversity 2011–2020 and to renewing nature and cultures. A complement to the fifth edition of Global Biodiversity Outlook*. Moreton-in-Marsh, England. https://www.cbd.int/gbo/gbo5/publication/lbo-2-en.pdf

Forest Peoples Programme, The International Indigenous Forum on Biodiversity, & The Secretariat of the Convention on Biological Diversity. (2016). *Local Biodiversity Outlooks. Indigenous Peoples' and Local Communities' Contributions to the Implementation of* *the Strategic Plan for Biodiversity 2011-2020. A complement to the fourth edition of the Global Biodiversity Outlook*. http://www.forestpeoples.org/sites/default/files/publication/2016/12/lbo-summary-2016-english-design-v6.pdf

Foster, J. D., Ellis, A. G., Foxcroft, L. C., Carroll, S. P., & Roux, J. L. (2019). The potential evolutionary impact of invasive balloon vines on native soapberry bugs in South Africa. *NeoBiota*, *49*, 19–35. https://doi.org/10.3897/neobiota.49.34245

Foxcroft, L. C., Pyšek, P., Richardson, D. M., & Genovesi, P. (Eds.). (2013). *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7

Foxcroft, L. C., Richardson, D. M., Pyšek, P., & Genovesi, P. (2013). Invasive alien plants in protected areas: Threats, opportunities, and the way forward. In *Plant Invasions in Protected Areas* (Vol. 7, pp. 621–639). Springer Netherlands. http://link.springer.com/10.1007/978-94-007-7750-7

Francour, P., Pellissier, V., Mangialajo, L., Buisson, E., Stadelmann, B., Veillard, N., Meinesz, A., Thibaut, T., & De Vaugelas, J. (2009). Changes in invertebrate assemblages of *Posidonia oceanica* beds following *Caulerpa taxifolia* invasion. *Vie et Milieu / Life & Environment*, *59*(1), 31–38. https://hal.archives-ouvertes.fr/hal-00927249

Frank, A. S., Johnson, C. N., Potts, J. M., Fisher, A., Lawes, M. J., Woinarski, J. C., Tuft, K., Radford, I. J., Gordon, I. J., & Collis, M.-A. (2014). Experimental evidence that feral cats cause local extirpation of small mammals in Australia's tropical savannas. *Journal of Applied Ecology*, *51*(6), 1486–1493. https://doi.org/10.1111/1365-2664.12323

Frelich, L. E., Hale, C. M., Scheu, S., Holdsworth, A. R., Heneghan, L., Bohlen, P. J., & Reich, P. B. (2006). Earthworm invasion into previously earthworm-free temperate and boreal forests. *Biological Invasions*, *8*(6), 1235–1245. https://doi.org/10.1007/s10530-006-9019-3

French, N. P. (2017). Impacts of non-native species on livestock. In *Impact of Biological Invasions on Ecosystem Services* (Vol. 12, pp. 139–154). Springer, Cham. https://link.springer.com/chapter/10.1007/978-3-319-45121-3_9

Frenot, Y., Chown, S. L., Whinam, J., Selkirk, P. M., Convey, P., Skotnicki, M., & Bergstrom, D. M. (2005). Biological invasions in the Antarctic: Extent, impacts and implications. *Biological Reviews*, *80*(1), 45–72. https://doi.org/10.1017/S1464793104006542

Frey, G., Emery, M. R., & Greenlaw, S. (2019). Weaving Together Livelihood and Culture in Maine, USA. In D. Pullanikkatil & C. M. Shackleton (Eds.), *Poverty Reduction Through Non-Timber Forest Products* (pp. 147–150). Springer International Publishing. https://doi.org/10.1007/978-3-319-75580-9_24

Frick, W. F., Kingston, T., & Flanders, J. (2020). A review of the major threats and challenges to global bat conservation. *Annals of the New York Academy of Sciences*, *1469*(1), 5–25. https://doi.org/10.1111/nyas.14045

Fried, G., Chauvel, B., Reynaud, P., & Sache, I. (2017). Decreases in crop production by non-native weeds, pests, and pathogens. In *Impact of biological invasions on ecosystem services* (Vol. 12, pp. 83–101). Springer. https://link.springer.com/chapter/10.1007/978-3-319-45121-3_6

Fritts, T. H. (2002). Economic costs of electrical system instability and power outages caused by snakes on the Island of Guam. *International Biodeterioration & Biodegradation*, *49*(2–3), 93–100. https://doi.org/10.1016/S0964-8305(01)00108-1

Fulton, E. A., Smith, A. D. M., & Johnson, C. R. (2003). Effect of complexity on marine ecosystem models. *Marine Ecology Progress Series*, *253*, 1–16. https://www.int-res.com/abstracts/meps/v253/p1-16

Fumanal, B., Chauvel, B., & Bretagnolle, F. (2007). Estimation of pollen and seed production of common ragweed in France. *Annals of Agricultural and Environmental Medicine*, *14*(2), 233–236.

Gabrio, T., Alberternst, B., Böhme, M., Kaminski, U., Nawrath, S., & Behrendt, H. (2010). Sensitization to allergens of *Ambrosia artemisiifolia* and other allergens on 10 years old children and adults in Baden-Württemberg. *Umweltmedizin in Forschung Und Praxis*, *15*(1), 15–22. https://www.cabdirect.org/cabdirect/abstract/20103075413

Gaiarin, B., & Durham, W. (2016). *The Beavers that Dam the End of the World: An Analysis of the Impact of Castor canadensis in Tierra del Fuego and Eradication Efforts*. Stanford University. https://vdocuments.mx/the-beavers-that-dam-the-end-of-the-world-an-analysis-of-introduction-the.html?page=1

Galanidi, M., Zenetos, A., & Bacher, S. (2018). Assessing the socio-economic impacts of priority marine invasive fishes in the Mediterranean with the newly proposed SEICAT methodology. *Mediterranean Marine Science*, *19*(1), 107–123. https://doi.org/10.12681/mms.15940

Galbreath, R., Onewhero, R. D., & Brown, D. (2004). The tale of the lighthouse-keeper's cat: Discovery and extinction of the Stephens Island wren (*Traversia lyalli*). *Notornis*, *51*(4), 193–200. https://www.researchgate.net/publication/285819081_The_tale_of_the_lighthouse-keeper's_cat_Discovery_and_extinction_of_the_Stephens_Island_wren_Traversia_lyalli

Galil, B. S. (2007). Loss or gain? Invasive aliens and biodiversity in the Mediterranean Sea. *Marine Pollution Bulletin*, *55*(7–9), 314–322. https://doi.org/10.1016/j.marpolbul.2006.11.008

Galil, B. S. (2017). Eyes Wide Shut: Managing Bio-Invasions in Mediterranean Marine Protected Areas. In P. D. Goriup (Ed.), *Management of Marine Protected Areas* (pp. 187–206). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781119075806.ch10

Galil, B. S. (2018). Poisonous and Venomous: Marine Alien Species in the Mediterranean Sea and Human Health. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health*. CABI.

Galil, B. S., Boero, F., Campbell, M. L., Carlton, J. T., Cook, E., Fraschetti, S., Gollasch, S., Hewitt, C. L., Jelmert, A., Macpherson, E., Marchini, A., McKenzie, C., Minchin, D., Occhipinti-Ambrogi, A., Ojaveer, H., Olenin, S., Piraino, S., & Ruiz, G. M. (2015). 'Double trouble': The expansion of the Suez Canal and marine bioinvasions in the Mediterranean Sea. *Biological Invasions*, *17*(4), 973–976. https://doi.org/10.1007/s10530-014-0778-y

Gallucci, F., Hutchings, P., Gribben, P., & Fonseca, G. (2012). Habitat alteration and community-level effects of an invasive ecosystem engineer: A case study along the coast of NSW, Australia. *Marine Ecology Progress Series*, *449*, 95–108. https://doi.org/10.3354/meps09547

Garbary, D. J., Miller, A. G., Seymour, N., & Williams, J. (2004). Destruction of eelgrass beds in Nova Scotia by the invasive green crab. *Status and Conservation of Eelgrass*, 13–14. http://www.bofep.org/PDFfiles/Eelgrass_Technica_%20Report.pdf#page=22

Gardner, J. P., Zbawicka, M., Westfall, K. M., & Wenne, R. (2016). Invasive blue mussels threaten regional scale genetic diversity in mainland and remote offshore locations: The need for baseline data and enhanced protection in the Southern Ocean. *Global Change Biology*, *22*(9), 3182–3195. https://doi.org/10.1111/gcb.13332

Gautam, M. R., Chief, K., & Smith, W. J. (2013). Climate change in arid lands and Native American socioeconomic vulnerability: The case of the Pyramid Lake Paiute Tribe. *Climatic Change*, *120*(3), 585–599. https://doi.org/10.1007/s10584-013-0737-0

Gehman, A.-L. M., & Byers, J. E. (2017). Non-native parasite enhances susceptibility of host to native predators. *Oecologia*, *183*(4), 919–926. https://doi.org/10.1007/s00442-016-3784-1

Gelder, S., & Williams, B. (2015). Global Overview of Branchiobdellida (Annelida: Clitellata). In T. Kawai, Z. Faulkes, & G. Scholtz (Eds.), *Freshwater Crayfish* (pp. 628–654). CRC Press. https://doi.org/10.1201/b18723-27

Geraldi, N. R., Anton, A., Lovelock, C. E., & Duarte, C. M. (2019). Are the ecological effects of the "worst" marine invasive species linked with scientific and media attention? *PLoS ONE*, *14*(4), e0215691. https://doi.org/10.1371/journal.pone.0215691

Gerlach, J. (2016). *Icons of evolution: Pacific island tree-snails of the family Partulidae*. Phelsuma Press.

Gerlach, J., Barker, G. M., Bick, C. S., Bouchet, P., Brodie, G., Christensen, C. C., Collins, T., Coote, T., Cowie, R. H., & Fiedler, G. C. (2021). Negative impacts of invasive predators used as biological control agents against the pest snail *Lissachatina fulica*: The snail *Euglandina 'rosea'*and the flatworm *Platydemus manokwari*. *Biological Invasions*, *23*(4), 997–1031. https://doi.org/10.1007/s10530-020-02436-w

Gherardi, F. (2010). Invasive crayfish and freshwater fishes of the world. *Revue Scientifique Et Technique (International Office of Epizootics)*, *29*(2), 241–254. https://doi.org/10.20506/rst.29.2.1973

Gherardi, F. (2011). Towards a sustainable human use of freshwater crayfish (Crustacea, Decapoda, Astacidea). *Knowledge and Management of Aquatic Ecosystems*, *401*, 02. https://doi.org/10.1051/kmae/2011038

Gichuru, N., Nyamweya, C., Owili, M., Mboya, D., & Wanyama, R. (2018). Poor management of Lake Victoria fisheries (Kenya); a threat to sustainable fish supplies. *Nature & Faune Journal*, *32*(2), 38–43. https://www.cabdirect.org/cabdirect/abstract/20193341327

Giovas, C. M. (2006). No pig atoll: Island biogeography and the extirpation of a Polynesian domesticate. *Asian Perspectives*, *45*(1), 69–95. https://www.jstor.org/stable/42928675

Girsang, S. S., Nurzannah, S. E., Girsang, M. A., & Effendi, R. (2020). The distribution and impact of fall army worm (*Spodoptera frugiperda*) on maize production in North Sumatera. *IOP Conference Series: Earth and Environmental Science*, *484*(1), 012099. https://doi.org/10.1088/1755-1315/484/1/012099

GISD. (2013). *100 of the World's Worst Invasive Alien Species*. Global Invasive Species Database. http://www.iucngisd.org/gisd/100_worst.php

GISP. (2005). El programa Mundial sobre Especies Invasoras (GISP). *Secretaría GISP*, *80*.

Glasby, T. M. (2013). *Caulerpa taxifolia* in seagrass meadows: Killer or opportunistic weed? *Biological Invasions*, *15*(5), 1017–1035. https://link.springer.com/article/10.1007/s10530-012-0347-1

Glatstein, M., Adir, D., Galil, B., Scolnik, D., Rimon, A., Pivko-Levy, D., & Hoyte, C. (2018). Pediatric jellyfish envenomation in the Mediterranean Sea. *European Journal of Emergency Medicine*, *25*(6), 434–439. https://doi.org/10.1097/MEJ.0000000000000479

Glisson, W. J., & Larkin, D. J. (2021). Hybrid watermilfoil (*Myriophyllum spicatum × Myriophyllum sibiricum*) exhibits traits associated with greater invasiveness than its introduced and native parental taxa. *Biological Invasions*, *23*(8), 2417–2433. https://doi.org/10.1007/s10530-021-02514-7

Global Invasive Species Database. (2010, October 4). *Species profile: Ulex europaeus*. http://www.iucngisd.org/gisd/speciesname/Ulex+europaeus

Goka, K. (2010). Introduction to the Special Feature for Ecological Risk Assessment of Introduced Bumblebees: Status of the European bumblebee, *Bombus terrestris*, in Japan as a beneficial pollinator and an invasive alien species. *Applied Entomology and Zoology*, *45*(1), 1–6. https://doi.org/10.1303/aez.2010.1

Gopi, K. C., & Radhakrishnan, C. (2002). Impact assessment of African Catfish (*Clarias gariepinus*) infestation on indigenous fish diversity in Manalur Grama Panchayat, Thrissur District, Kerala: A case study. *ENVIS Newsletter, Zoological Survey of India*, *9*(1–2), 9–12.

Gorokhova, E., Fagerberg, T., & Hansson, S. (2004). Predation by herring (*Clupea harengus*) and sprat (*Sprattus sprattus*) on *Cercopagis pengoi* in a western Baltic Sea bay. *ICES Journal of Marine Science*, *61*(6), 959–965. https://doi.org/10.1016/j.icesjms.2004.06.016

Goudswaard, K. P., Witte, F., & Katunzi, E. F. (2008). The invasion of an introduced predator, Nile perch (*Lates niloticus*, L.) in Lake Victoria (East Africa): Chronology and causes. *Environmental Biology of Fishes*, *81*(2), 127–139. https://doi.org/10.1007/s10641-006-9180-7

Gozlan, R. E. (2017). Interference of non-native species with fisheries and aquaculture. In *Impact of biological invasions on ecosystem services* (pp. 119–137). Springer. https://link.springer.com/chapter/10.1007/978-3-319-45121-3_8

Gozlan, R. E., & Newton, A. C. (2009). Biological Invasions: Benefits *versus* Risks. *Science*, *324*(5930), 1015–1015. https://doi.org/10.1126/science.324_1015a

Gray, A. J., Marshall, D. F., & Raybould, A. F. (1991). A Century of Evolution in *Spartina anglica*. *Advances in Ecological Research*, *21*(C), 1–62. https://doi.org/10.1016/S0065-2504(08)60096-3

Gregory, G. J., & Quijón, P. A. (2011). The impact of a coastal invasive predator on infaunal communities: Assessing the roles of density and a native counterpart. *Journal of Sea Research*, *66*(2), 181–186. https://doi.org/10.1016/j.seares.2011.05.009

Gremmen, N. J. M., Chown, S. L., & Marshall, D. J. (1998). Impact of the introduced grass *Agrostis stolonifera* on vegetation and soil fauna communities at Marion Island, sub-Antarctic. *Biological Conservation*, *85*(3), 223–231. https://doi.org/10.1016/S0006-3207(97)00178-X

Grewell, B. J., Netherland, M. D., & Skaer Thomason, M. J. (2016). *Establishing research and management priorities for invasive water primroses (*Ludwigia *spp.)* (ERDC/EL TR-16-2; p. 55). The U.S. Army Engineer Research and Development Center (ERDC). https://erdc-library.erdc.

dren.mil/jspui/bitstream/11681/10110/1/ERDC-EL%20TR-16-2%20Revised.pdf

Gribben, P. E., Wright, J. T., O'Connor, W. A., Doblin, M. A., Eyre, B., & Steinberg, P. D. (2009). Reduced Performance of Native Infauna Following Recruitment to a Habitat-Forming Invasive Marine Alga. *Oecologia*, *158*(4), 733–745. https://www.jstor.org/stable/40309791

Griffiths, C. L., Hockey, P. A. R., Van Erkom Schurink, C., & Le Roux, P. J. (1992). Marine invasive aliens on South African shores: Implications for community structure and trophic functioning. *South African Journal of Marine Science*, *12*(1), 713–722. https://doi.org/10.2989/02577619209504736

Grigg, R. W. (2003). Invasion of a deep black coral bed by an alien species, *Carijoa riisei*, off Maui, Hawaii. *Coral Reefs*, *22*(2), 121–122. https://doi.org/10.1007/s00338-003-0306-5

Grosholz, E. D., Ruiz, G. M., Dean, C. A., Shirley, K. A., Maron, J. L., & Connors, P. G. (2000). The impacts of a nonindigenous marine predator in a California Bay. *Ecology*, *81*(5), 1206–1224. https://esajournals.onlinelibrary.wiley.com/doi/abs/10.1890/0012-9658(2000)081[1206:TIOANM]2.0.CO;2

Gruber, M. A. M., Santoro, D., Cooling, M., Lester, P. J., Hoffmann, B. D., Boser, C., & Lach, L. (2022). A global review of socioeconomic and environmental impacts of ants reveals new insights for risk assessment. *Ecological Applications*, *32*(4). https://doi.org/10.1002/eap.2577

Gruner, D. (2004). *Arthropods from ʻōhiʻa Lehua (Myrtaceae: Metrosideros polymorpha), With New Records for the Hawaiian Islands*. *78*, 33–52. https://drum.lib.umd.edu/handle/1903/7109

Guerra, N. M., Leite, A. P., Souza, A. dos S., Ribeiro, J. E. da S., Ribeiro, J. P. de O., Oliveira, R. S. de, Alves, C. A. B., de Sousa Júnior, S. P., Lima, J. R. de F., & de Lucena, R. F. P. (2014). Uso de algaroba (*Prosopis juliflora* (S.W.) DC) en las comunidades tradicionales de las regiones semiáridas del Nordeste de Brasil. *Gaia Scientia*, 124–136. http://periodicos.ufpb.br/ojs2/index.php/gaia/index

Gutrich, J. J., VanGelder, E., & Loope, L. (2007). Potential economic impact of introduction and spread of the red imported fire ant, *Solenopsis invicta*, in Hawaii. *Environmental Science & Policy*, *10*(7–8), 685–696. https://doi.org/10.1016/j.envsci.2007.03.007

Guzy, J. C., Falk, B. G., Smith, B. J., Willson, J. D., Reed, R. N., Aumen, N. G., Avery, M. L., Bartoszek, I. A., Campbell, E., Cherkiss, M. S., Claunch, N. M., Currylow, A. F., Dean, T., Dixon, J., Engeman, R., Funck, S., Gibble, R., Hengstebeck, K. C., Humphrey, J. S., … Hart, K. M. (2023). Burmese pythons in Florida: A synthesis of biology, impacts, and management tools. *NeoBiota*, *80*, 1–119. https://doi.org/10.3897/neobiota.80.90439

Haack, R. A., Jendak, E., Houping, L., Marchant, K. R., Petrice, T. R., Poland, T. M., & Ye, H. (2002). The emerald ash borer: A new exotic pest in North America. *Michigan Entomological Society*, *47*(3 & 4), 2. https://www.researchgate.net/publication/228544207_The_Emerald_Ash_Borer_A_New_Exotic_Pest_in_North_America

Hamer, H. H., Malzahn, A. M., & Boersma, M. (2011). The invasive ctenophore *Mnemiopsis leidyi*: A threat to fish recruitment in the North Sea? *Journal of Plankton Research*, *33*(1), 137–144. https://doi.org/10.1093/plankt/fbq100

Han, B. A., Kramer, A. M., & Drake, J. M. (2016). Global Patterns of Zoonotic Disease in Mammals. *Trends in Parasitology*, *32*(7), 565–577. https://doi.org/10.1016/j.pt.2016.04.007

Hanekom, N. (2008). Invasion of an indigenous *Perna perna* mussel bed on the south coast of South Africa by an alien mussel *Mytilus galloprovincialis* and its effect on the associated fauna. *Biological Invasions*, *10*(2), 233–244. https://doi.org/10.1007/s10530-007-9125-x

Hanekom, N., & Nel, P. (2002). Invasion of sandflats in Langebaan Lagoon, South Africa, by the alien mussel *Mytilus galloprovincialis*: Size, composition and decline of the populations. *African Zoology*, *37*(2), 197–208. https://doi.org/10.1080/15627020.2002.11657175

Harmelin-Vivien, M., Harmelin, J.-G., & Francour, P. (1996). A 3-year study of the littoral fish fauna of sites colonized by *Caulerpa taxifolia* in the NW Mediterranean (Menton, France). *Second International Workshop on Caulerpa Taxifolia*, 391–397. https://www.researchgate.net/publication/235046447_A_3-year_study_of_the_littoral_fish_fauna_of_sites_colonized_by_*Caulerpa*_taxifolia_in_the_NW_Mediterranean_Menton_France

Harris, P. (1984). Biocontrol of Weeds: Beaurocrats, Botanists, Beekeepers and Other Bottlenecks. *Proc. VI Int. Symp. Biol. Contr. Wecds*, *19*, 25.

Haubrock, P. J., Bernery, C., Cuthbert, R. N., Liu, C., Kourantidou, M., Leroy, B., Turbelin, A. J., Kramer, A. M., Verbrugge, L. N. H., Diagne, C., Courchamp, F., & Gozlan, R. E. (2022). Knowledge gaps in economic costs of invasive alien fish worldwide. *Science of The Total Environment*, *803*, 149875. https://doi.org/10.1016/j.scitotenv.2021.149875

Haubrock, P. J., Cuthberg, R. N., Ricciardi, A., Diagne, C., & Courchamp, F. (2021). Massive economic costs of invasive bivalves in freshwater ecosystems. *Research Square*, 22. https://doi.org/10.21203/rs.3.rs-389696/v1

Haubrock, P. J., Cuthbert, R. N., Sundermann, A., Diagne, C., Golivets, M., & Courchamp, F. (2021). Economic costs of invasive species in Germany. *NeoBiota*, *67*, 225–246. https://doi.org/10.3897/neobiota.67.59502

Haubrock, P. J., Cuthbert, R. N., Yeo, D. C. J., Banerjee, A. K., Liu, C., Diagne, C., & Courchamp, F. (2021). Biological invasions in Singapore and Southeast Asia: Data gaps fail to mask potentially massive economic costs. *NeoBiota*, *67*, 131–152. https://doi.org/10.3897/neobiota.67.64560

Haubrock, P. J., Turbelin, A. J., Cuthbert, R. N., Novoa, A., Taylor, N. G., Angulo, E., Ballesteros-Mejia, L., Bodey, T. W., Capinha, C., Diagne, C., Essl, F., Golivets, M., Kirichenko, N., Kourantidou, M., Leroy, B., Renault, D., Verbrugge, L., & Courchamp, F. (2021). Economic costs of invasive alien species across Europe. *NeoBiota*, *67*, 153–190. https://doi.org/10.3897/neobiota.67.58196

Haussmann, N. S., Rudolph, E. M., Kalwij, J. M., & McIntyre, T. (2013). Fur seal populations facilitate establishment of exotic vascular plants. *Biological Conservation*, *162*, 33–40. https://doi.org/10.1016/j.biocon.2013.03.024

Hayes, R. A., Crossland, M. R., Hagman, M., Capon, R. J., & Shine, R. (2009). Ontogenetic variation in the chemical defenses of cane toads (*Bufo marinus*): Toxin profiles and effects on predators. *Journal of Chemical Ecology*, *35*(4), 391–399. https://doi.org/10.1007/s10886-009-9608-6

Head, L., & Atchison, J. (2015). Entangled invasive lives: Indigenous invasive plant management in northern australia. *Geografiska Annaler: Series B, Human Geography*, *97*(2), 169–182. https://doi.org/10.1111/geob.12072

Heath, D. D., Rawson, P. D., & Hilbish, T. J. (1995). PCR-based nuclear markers identify alien blue mussel (*Mytilus* spp.) genotypes on the west coast of Canada. *Canadian Journal of Fisheries and Aquatic Sciences*, *52*(12), 2621–2627. https://doi.org/10.1139/f95-851

Heger, T., Bernard-Verdier, M., Gessler, A., Greenwood, A. D., Grossart, H.-P., Hilker, M., Keinath, S., Kowarik, I., Kueffer, C., Marquard, E., Müller, J., Niemeier, S., Onandia, G., Petermann, J. S., Rillig, M. C., Rödel, M.-O., Saul, W.-C., Schittko, C., Tockner, K., … Jeschke, J. M. (2019). Towards an Integrative, Eco-Evolutionary Understanding of Ecological Novelty: Studying and Communicating Interlinked Effects of Global Change. *BioScience*, *69*(11), 888–899. https://doi.org/10.1093/biosci/biz095

Heller, W. P., Hughes, M. A., Luiz, B. C., Brill, E., Friday, J. B., Williams, A. M., & Keith, L. M. (2019). First report of *Ceratocystis huliohia* causing mortality of *Metrosideros polymorpha* trees on the Island of Kauaʻi, Hawaiʻi USA. *Forest Pathology*, *49*(5), e12546. https://doi.org/10.1111/efp.12546

Hempel, M., Neukamm, R., & Thiel, R. (2016). Effects of introduced round goby (*Neogobius melanostomus*) on diet composition and growth of zander (*Sander lucioperca*), a main predator in European brackish waters. *Aquatic Invasions*, *11*(2), 167–178. https://doi.org/10.3391/ai.2016.11.2.06

Herbert, R. J. H., Davies, C. J., Bowgen, K. M., Hatton, J., & Stillman, R. A. (2018). The importance of nonnative Pacific oyster reefs as supplementary feeding areas for coastal birds on estuary mudflats. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *28*(6), 1294–1307. https://doi.org/10.1002/aqc.2938

Heringer, G., Angulo, E., Ballesteros-Mejia, L., Capinha, C., Courchamp, F., Diagne, C., Duboscq-Carra, V. G., Nuñez, M. A., & Zenni, R. D. (2021). The economic costs of biological invasions in Central and South America: A first regional assessment. *NeoBiota*, *67*, 401–426. https://doi.org/10.3897/neobiota.67.59193

Heringer, G., Thiele, J., Meira-Neto, J. A. A., & Neri, A. V. (2019). Biological invasion threatens the sandy-savanna *Mussununga* ecosystem in the Brazilian Atlantic Forest. *Biological Invasions*, *21*(6), 2045–2057. https://doi.org/10.1007/s10530-019-01955-5

Herms, D. A., & McCullough, D. G. (2014). Emerald Ash Borer Invasion of North America: History, Biology, Ecology, Impacts, and Management. *Annual Review of Entomology*, *59*(1), 13–30. https://doi.org/10.1146/annurev-ento-011613-162051

Heukelbach, J., Alencar, C. H., Kelvin, A. A., Oliveira, W. K. de, & Cavalcanti, L. P. de G. (2016). Zika virus outbreak in Brazil. *The Journal of Infection in Developing Countries*, *10*(02), Article 02. https://doi.org/10.3855/jidc.8217

Hill, M. P., Coetzee, J. A., Martin, G. D., Smith, R., & Strange, E. F. (2020). Invasive alien aquatic plants in South African freshwater ecosystems. In B. van Wilgen, J. Measey, D. Richardson, J. Wilson, & T. Zengeya (Eds.), *Biological Invasions in South Africa* (pp. 97–114). Springer, Cham. https://doi.org/10.1007/978-3-030-32394-3_4

Hinderaker, S. E., & Nielsen, A. (2022). Current status of important nature values in the Vega archipelago. In *38* (8(36); NIBIO Rapport). NIBIO. https://nibio.brage.unit.no/nibio-xmlui/handle/11250/3021612

Hines, A. H., Alvarez, F., & Reed, S. A. (1997). Introduced and native populations of a marine parasitic castrator: Variation in prevalence of the Rhizocephalan *Loxothylacus panopaei* in xanthid crabs. *Bulletin of Marine Science*, *61*(2), 197–214. http://repository.si.edu/xmlui/handle/10088/17860

Hiremath, A. J., & Sundaram, B. (2005). The fire-lantana cycle hypothesis in Indian forests. *Conservation and Society*, *3*(1), 26–42. JSTOR. https://www.jstor.org/stable/26396598

Hirsch, H., Allsopp, M. H., Canavan, S., Cheek, M., Geerts, S., Geldenhuys, C. J., Harding, G., Hurley, B. P., Jones, W., Keet, J.-H., Klein, H., Ruwanza, S., van Wilgen, B. W., Wingfield, M. J., & Richardson, D. M. (2020). *Eucalyptus camaldulensis* in South Africa – past, present, future. *Transactions of the Royal Society of South Africa*, *75*(1), 1–22. https://doi.org/10.1080/0035919X.2019.1669732

Hoagland, P., & Jin, D. (2006). Science and Economics in the Management of an Invasive Species. *BioScience*, *56*(11), 931–935. https://doi.org/10.1641/0006-3568(2006)56[931:SAEITM]2.0.CO;2

Hockey, P. A. R., & van Erkom Schurink, C. (1992). The invasive biology of the mussel *Mytilus galloprovincialis* on the southern African coast. *Transactions of the Royal Society of South Africa*, *48*(1), 123–139. https://doi.org/10.1080/00359199209520258

Hoffmann, B. D., & Broadhurst, L. M. (2016). The economic cost of managing invasive species in Australia. *NeoBiota*, *31*, 1–18. https://doi.org/10.3897/neobiota.31.6960

Holdich, D. M., Reynolds, J. D., Souty-Grosset, C., & Sibley, P. J. (2009). A review of the ever increasing threat to European crayfish from non-indigenous crayfish species. *Knowledge and Management of Aquatic Ecosystems*, *394–395*, 11. https://doi.org/10.1051/kmae/2009025

Holdsworth, A. R., Frelich, L. E., & Reich, P. B. (2007). Effects of Earthworm Invasion on Plant Species Richness in Northern Hardwood Forests. *Conservation Biology*, *21*(4), 997–1008. https://doi.org/10.1111/j.1523-1739.2007.00740.x

Holmes, M. C. C., & Jampijinpa, W. (Stephen P. (2013). Law for Country: The Structure of Warlpiri Ecological Knowledge and Its Application to Natural Resource Management and Ecosystem Stewardship. *Ecology and Society*, *18*(3), art19. https://doi.org/10.5751/ES-05537-180319

Holmes, T. P., Aukema, J. E., Von Holle, B., Liebhold, A., & Sills, E. (2009). Economic Impacts of Invasive Species in Forests: Past, Present, and Future. *Annals of the New York Academy of Sciences*, *1162*(1), 18–38. https://doi.org/10.1111/j.1749-6632.2009.04446.x

Houghton, M., Terauds, A., Merritt, D., Driessen, M., & Shaw, J. (2019). The impacts of non-native species on the invertebrates of Southern Ocean Islands. *Journal of Insect Conservation*, *23*(3), 435–452. https://doi.org/10.1007/s10841-019-00147-9

Houngbo, S., Zannou, A., Aoudji, A., Sossou, H. C., Sinzogan, A., Sikirou, R., Zossou, E., Totin Vodounon, H. S., Adomou, A., & Ahanchédé, A. (2020). Farmers' knowledge and management practices of fall armyworm, *Spodoptera frugiperda* (J.E. Smith) in Benin, West Africa. *Agriculture (Switzerland)*, *10*(10), 1–15. https://doi.org/10.3390/agriculture10100430

Howard, B. R., Francis, F. T., Côté, I. M., & Therriault, T. W. (2019). Habitat alteration by invasive European green crab (*Carcinus maenas*) causes eelgrass loss in British Columbia, Canada. *Biological Invasions*, *21*(12), 3607–3618. https://doi.org/10.1007/s10530-019-02072-z

Howard, P. L. (2019). Human adaptation to invasive species: A conceptual framework based on a case study metasynthesis. *Ambio*, *48*(12), 1401–1430. https://doi.org/10.1007/s13280-019-01297-5

Hoyos, L. E., Gavier-Pizarro, G. I., Kuemmerle, T., Bucher, E. H., Radeloff, V. C., & Tecco, P. A. (2010). Invasion of glossy privet (*Ligustrum lucidum*) and native forest loss in the Sierras Chicas of Córdoba, Argentina. *Biological Invasions*, *12*(9), 3261–3275. https://doi.org/10.1007/s10530-010-9720-0

Hu, Y., Gillespie, G., & Jessop, T. S. (2019). Variable reptile responses to introduced predator control in southern Australia. *Wildlife Research*, *46*(1), 64–75. https://doi.org/10.1071/WR18047

Hughes, K. A., Pertierra, L. R., Molina-Montenegro, M. A., & Convey, P. (2015). Biological invasions in terrestrial Antarctica: What is the current status and can we respond? *Biodiversity and Conservation*, *24*(5), 1031–1055. https://doi.org/10.1007/s10531-015-0896-6

Hughes, K. A., Pescott, O. L., Peyton, J. M., Adriaens, T., Cottier-Cook, E. J., Key, G., Rabitsch, W., Tricarico, E., Barnes, D. K. A., Baxter, N., Belchier, M., Blake, D., Convey, P., Dawson, W., Frohlich, D., Gardiner, L. M., González-Moreno, P., James, R., Malumphy, C., … Roy, H. E. (2020). Invasive non-native species likely to threaten biodiversity and ecosystems in the Antarctic Peninsula region. *Global Change Biology*, *26*(4), 2702–2716. https://doi.org/10.1111/gcb.14938

Hulme, P. E. (2014). Invasive species challenge the global response to emerging diseases. *Trends in Parasitology*, *30*(6), 267–270. https://doi.org/10.1016/j.pt.2014.03.005

Hunter, D. O., Lagisz, M., Leo, V., Nakagawa, S., & Letnic, M. (2018). Not all predators are equal: A continent-scale analysis of the effects of predator control on Australian mammals. *Mammal Review*, *48*(2), 108–122. https://doi.org/10.1111/mam.12115

Imada, C. T. (2012). *Hawaiian native and naturalized vascular plants checklist* (Bishop Musem Technical Report 60; p. 380). Bishop Museum. http://www.aecos.com/AECOS/EG%20Files/eg2012002.pdf

Ingeman, K. E. (2016). Lionfish cause increased mortality rates and drive local extirpation of native prey. *Marine Ecology Progress Series*, *558*, 235–245. https://doi.org/10.3354/meps11821

Innes, J., Kelly, D., Overton, J. M., & Gillies, C. (2010). Predation and other factors currently limiting New Zealand forest birds. *New Zealand Journal of Ecology*, *34*(1), 86. http://www.newzealandecology.org/nzje/

IPBES. (2016). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3235428

IPBES. (2018a). *The IPBES regional assessment report on biodiversity and ecosystem services for Asia and the Pacific* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237374

IPBES. (2018b). *The IPBES regional assessment report on biodiversity and ecosystem services for Europe and Central Asia* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237429

IPBES. (2019). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3831673

IPBES. (2020). *Report of the Indigenous and local knowledge dialogue workshop on the first order draft of the IPBES assessment of invasive alien species* (J. L. Andreve, R. Batzin Chojoj, A. Black, J. T. Cleofe, F. Daguitan, C. Grant, J. A. Guillao, L. Jacobs, T. Malcolm, L. Mulenkei, K. Kumar Rai, A. Nzovu, J. M. Ole Kaunga, M. E. Regpala, N. Sall, P. Shulbaeva, R. Spencer, P. Timoti, & Y. Upun, Eds.). https://ipbes.net/sites/default/files/inline-files/IPBES_IAS_2ndILKDialogue_FOD_Report_FINAL_ForWeb.pdf

IPBES. (2022). *Report of the third Indigenous and local knowledge dialogue workshop for the IPBES thematic assessment of invasive alien species and their control.* https://ipbes.net/sites/default/files/2023-02/IPBES_IAS_3rdILKDialogue_SPM-SOD_Report_FinalForWeb2.pdf

IPCC. (2014). *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects.* (C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White, Eds.; p. 1132). Cambridge University Press. https://doi.org/10.1017/CBO9781107415379

Irigoyen, A. J., Trobbiani, G., Sgarlatta, M. P., & Raffo, M. P. (2011). Effects of the alien algae *Undaria pinnatifida* (Phaeophyceae, Laminariales) on the diversity and abundance of benthic macrofauna in Golfo Nuevo (Patagonia, Argentina): Potential implications for local food webs. *Biological Invasions*, *13*(7), 1521–1532. https://doi.org/10.1007/s10530-010-9910-9

IUCN. (2020). *IUCN EICAT Categories and Criteria. The Environmental Impact Classification for Alien Taxa.* (First edition). IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2020.05.en

IUCN. (2022). *IUCN – CMP Unified Classification of Direct Threats. Version: 3.3*. https://www.iucnredlist.org/resources/threat-classification-scheme

IUCN. (2021). *The IUCN Red List of Threatened Species. Version 2021-3*. IUCN Red List of Threatened Species. https://www.iucnredlist.org/en

Jackson, J. E., Raadik, T. A., Lintermans, M., & Hammer, M. (2004). Alien salmonids in Australia: Impediments to effective impact management, and future directions. *New Zealand Journal of Marine and Freshwater Research*, *38*(3), 447–455. https://doi.org/10.1080/00288330.2004.9517252

Jackson, M. C., Loewen, C. J. G., Vinebrooke, R. D., & Chimimba, C. T. (2016). Net effects of multiple stressors in freshwater ecosystems: A meta-analysis. *Global Change Biology*, *22*(1), 180–189. https://doi.org/10.1111/gcb.13028

Jäger, S. (2000). Ragweed (Ambrosia) sensitisation rates correlate withthe amount of inhaled airborne pollen. A 14-year studyin Vienna, Austria. *Aerobiologia*, *16*(1), 149–153. https://doi.org/10.1023/A:1007603321556

Jaklič, M., Koren, Š., & Jogan, N. (2020). Alien water lettuce (*Pistia stratiotes* L.) outcompeted native macrophytes and altered the ecological conditions of a Sava oxbow lake (SE Slovenia). *Acta Botanica Croatica*, *79*(1), 0–0. https://doi.org/10.37427/botcro-2020-009

Jakubska-Busse, A., Śliwiński, M., & Kobyłka, M. (2013). Identification of bioactive components of essential oils in *Heracleum sosnowskyi* and *Heracleum mantegazzianum* (Apiaceae). *Archives of Biological Sciences*, *65*(3), 877–883. https://doi.org/10.2298/ABS1303877J

James, A. I., & Eldridge, D. J. (2007). Reintroduction of fossorial native mammals and potential impacts on ecosystem processes in an Australian desert landscape. *Biological Conservation*, *138*(3–4), 351–359. https://doi.org/10.1016/j.biocon.2007.04.029

James, A. I., Eldridge, D. J., Koen, T. B., & Moseby, K. E. (2011). Can the invasive European rabbit (*Oryctolagus cuniculus*) assume the soil engineering role of locally-extinct natives? *Biological Invasions*, *13*(12), 3027–3038. https://doi.org/10.1007/s10530-011-9987-9

Jarić, I., Heger, T., Castro Monzon, F., Jeschke, J. M., Kowarik, I., McConkey, K. R., Pyšek, P., Sagouis, A., & Essl, F. (2019). Crypticity in Biological Invasions. *Trends in Ecology & Evolution*, *34*(4), 291–302. https://doi.org/10.1016/j.tree.2018.12.008

Jaspers, C., Titelman, J., Hansson, L. J., Haraldsson, M., & Ditlefsen, C. R. (2011). The invasive ctenophore *Mnemiopsis leidyi* poses no direct threat to Baltic cod eggs and larva. *Limnology and Oceanography*, *56*(2), 431–439. https://doi.org/10.4319/lo.2011.56.2.0431

Javidpour, J., Molinero, J. C., Lehmann, A., Hansen, T., & Sommer, U. (2009). Annual assessment of the predation of *Mnemiopsis leidyi* in a new invaded environment, the Kiel Fjord (Western Baltic Sea): A matter of concern? *Journal of Plankton Research*, *31*(7), 729–738. https://doi.org/10.1093/plankt/fbp021

Jean Desbiez, A. L., Keuroghlian, A., Piovezan, U., & Bodmer, R. E. (2011). Invasive species and bushmeat hunting contributing to wildlife conservation: The case of feral pigs in a Neotropical wetland. *Oryx*, *45*(1), 78–83. https://doi.org/10.1017/S0030605310001304

Jemal, A., & Hugh-Jones, M. (1993). A review of the red imported fire ant (*Solenopsis invicta* Buren) and its impacts on plant, animal, and human health. *Preventive Veterinary Medicine*, *17*(1–2), 19–32. https://doi.org/10.1016/0167-5877(93)90051-T

Jeschke, J. M., Bacher, S., Blackburn, T. M., Dick, J. T. A., Essl, F., Evans, T., Gaertner, M., Hulme, P. E., Kühn, I., Mrugała, A., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D. M., Sendek, A., Vilà, M., Winter, M., & Kumschick, S. (2014). Defining the Impact of Non-Native Species: Impact of Non-Native Species. *Conservation Biology*, *28*(5), 1188–1194. https://doi.org/10.1111/cobi.12299

Jevon, T., & Shackleton, C. M. (2015). Integrating Local Knowledge and Forest Surveys to Assess *Lantana camara* Impacts on Indigenous Species Recruitment in Mazeppa Bay, South Africa. *Human Ecology*, *43*(2), 247–254. https://doi.org/10.1007/s10745-015-9748-y

Johnston, M. W., Bernard, A. M., & Shivji, M. S. (2017). Forecasting lionfish sources and sinks in the Atlantic: Are Gulf of Mexico reef fisheries at risk? *Coral Reefs*, *36*(1), 169–181. https://doi.org/10.1007/s00338-016-1511-3

Johnstone, R. W., & Olafsson, E. (1995). Some environmental aspects of open water algal cultivation: Zanzibar, Tanzania. *Ambio (Sweden)*. https://scholar.google.com/scholar_lookup?title=Some+environmental+aspects+of+open+water+algal+cultivation%3A+Zanzibar%2C+Tanzania&author=Johnstone%2C+R.W.&publication_year=1995

Jones, B. A. (2016). Work more and play less? Time use impacts of changing ecosystem services: The case of the invasive emerald ash borer. *Ecological Economics*, *124*, 49–58. https://doi.org/10.1016/j.ecolecon.2016.02.003

Jones, B. A. (2017). Invasive Species Impacts on Human Well-being Using the Life Satisfaction Index. *Ecological Economics*, *134*, 250–257. https://doi.org/10.1016/j.ecolecon.2017.01.002

Jones, B. A., & McDermott, S. M. (2018). Health Impacts of Invasive Species Through an Altered Natural Environment: Assessing Air Pollution Sinks as a Causal Pathway. *Environmental and Resource Economics*, *71*(1), 23–43. https://doi.org/10.1007/s10640-017-0135-6

Jones, C. W., Risi, M. M., Cleeland, J., & Ryan, P. G. (2019). First evidence of mouse attacks on adult albatrosses and petrels breeding on sub-Antarctic Marion and Gough Islands. *Polar Biology*, *42*, 619–623. https://doi.org/10.1007/s00300-018-02444-6

Jones, H. P., Tershy, B. R., Zavaleta, E. S., Croll, D. A., Keitt, B. S., Finkelstein, M. E., & Howald, G. R. (2008). Severity of the Effects of Invasive Rats on Seabirds: A Global Review. *Conservation Biology*, *22*(1), 16–26. https://doi.org/10.1111/j.1523-1739.2007.00859.x

Jones, J. M., White, I. R., White, J. M. L., & McFadden, J. P. (2009). Allergic contact dermatitis to English ivy (*Hedera helix*)—A case series. *Contact Dermatitis*, *60*(3), 179–180. https://doi.org/10.1111/j.1600-0536.2008.01492.x

Jones, M. G. W., & Ryan, P. G. (2010). Evidence of mouse attacks on albatross chicks on sub-Antarctic Marion Island. *Antarctic Science*, *22*(1), 39. https://doi.org/10.1017/S0954102009990459

Jones, P., & Closs, G. (2018). The introduction of brown trout to New Zealand and their impact on native fish communities. *Brown Trout: Biology, Ecology and Management*, 545–567. https://doi.org/10.1002/9781119268352.ch21

Joshi, P. N., Kumar, V., Koladiya, M., Patel, Y. S., & Karthik, T. (2009). Local perceptions of grassland change and priorities for conservation of natural resources of Banni, Gujarat, India. *Frontiers of Biology in China*, *4*(4), 549–556. https://doi.org/10.1007/s11515-009-0041-6

Joshi, R. C., Delacruz, M. S., Martin, E. C., Cabigat, J. C., Bahatan, R. G., Bahatan, A. D., Abayao, E. H., Choy-Awon, J., Chilagan, N. P., & Cayong, A. B. (2001). Current Status of the Golden Apple Snail in the Ifugao Rice Terraces, Philippines. *Journal of Sustainable Agriculture*, *18*(2–3), 71–90. https://doi.org/10.1300/J064v18n02_07

Jouventin, P., Bried, J., & Micol, T. (2003). Insular bird populations can be saved from rats: A long-term experimental study of white-chinned petrels *Procellaria aequinoctialis* on Ile de la Possession (Crozet archipelago). *Polar Biology*, *26*(6), 371–378. https://doi.org/10.1007/s00300-003-0497-9

Jubase, N., Shackleton, R. T., & Measey, J. (2021). Public awareness and perceptions of invasive alien species in small towns. *Biology*, *10*(12), 1322. https://doi.org/10.3390/biology10121322

Juliano, S. A., & Lounibos, P. L. (2005). Ecology of invasive mosquitoes: Effects on resident species and on human health: Invasive mosquitoes. *Ecology Letters*, *8*(5), 558–574. https://doi.org/10.1111/j.1461-0248.2005.00755.x

Kahng, S. E. (2007). *Ecological impacts of Carijoa riisei on black coral habitat* (p. 5). Western Pacific Fisheries Management Council. https://www.wpcouncil.org/precious/Documents/Carijoa%20Report.pdf

Kahng, S. E., & Grigg, R. W. (2005). Impact of an alien octocoral, *Carijoa riisei*, on black corals in Hawaii. *Coral Reefs*, *24*(4), 556–562. https://doi.org/10.1007/s00338-005-0026-0

Kamalakannan, B., Jeevamani, J. J. J., Nagendran, N. A., Pandiaraja, D., & Chandrasekaran, S. (2014). Impact of removal of invasive species *Kappaphycus alvarezii* from coral reef ecosystem in Gulf of Mannar, India. *Current Science*, *106*(10), 8. https://www.jstor.org/stable/24102487

Kamalakannan, B., Jeevamani, J. J. J., Nagendran, N. A., Pandiaraja, D., Kutty, N. K., & Chandrasekaran, S. (2010). *Turbinaria* sp. As victims to *Kappaphycusalvarezii* in reefs of Gulf of Mannar, India. *Coral Reefs*, *29*(4), 1077–1077. https://doi.org/10.1007/s00338-010-0684-4

Kamburska, L., Doncheva, V., & Stefanova, K. (2003). On the recent changes of zooplankton community structure along the Bulgarian Black Sea coast–a post invasion effect of exotic ctenophores interactions. *Proceedings of the First International Conference on ICES Cooperative Research Report*, *300*, 75. https://www.researchgate.net/publication/288956076_On_the_recent_changes_of_zooplankton_community_structure_along_the_bulgarian_black_sea_coast-A_post_invasion_effect_of_exotic_ctenophores_interactions

Kannan, R., Shackleton, C. M., Shaanker, @bullet R Uma, Kannan, R., Shackleton, Á. C. M., Shaanker, Á. R. U., & Shaanker, R. U. (2014). Invasive alien species as drivers in socio-ecological systems: Local adaptations towards use of *Lantana* in Southern India. *Environment, Development and Sustainability*, *16*(3), 649–669. https://doi.org/10.1007/s10668-013-9500-y

Kansiime, M. K., Mugambi, I., Rwomushana, I., Nunda, W., Lamontagne-Godwin, J., Rware, H., Phiri, N. A., Chipabika, G., Ndlovu, M., & Day, R. (2019). Farmer perception of fall armyworm (*Spodoptera frugiderda* J.E. Smith) and farm-level management practices in Zambia. *Pest Management Science*, *75*(10), 2840–2850. https://doi.org/10.1002/ps.5504

Kaplan, K. A., Hart, D. R., Hopkins, K., Gallager, S., York, A., Taylor, R., & Sullivan, P. J. (2017). Evaluating the interaction of the invasive tunicate *Didemnum vexillum* with the Atlantic sea scallop *Placopecten magellanicus* on open and closed fishing grounds of Georges Bank. *ICES Journal of Marine Science*, *74*(9), 2470–2479. https://doi.org/10.1093/icesjms/fsx076

Karatayev, A. Y., Burlakova, L. E., & Padilla, D. K. (2005). Contrasting Distribution and Impacts of Two Freshwater Exotic Suspension Feeders, *Dreissena polymorpha* and *Corbicula fluminea*. In R. F. Dame & S. Olenin (Eds.), *The Comparative Roles of Suspension-Feeders in Ecosystems* (Vol. 47, pp. 239–262). Springer-Verlag. https://doi.org/10.1007/1-4020-3030-4_14

Kassie, M., Wossen, T., De Groote, H., Tefera, T., Sevgan, S., & Balew, S. (2020). Economic impacts of fall armyworm and its management strategies: Evidence from southern Ethiopia. *European Review of Agricultural Economics*, *47*(4), 1473–1501. https://doi.org/10.1093/erae/jbz048

Kasulo, K., & Perrings, C. (2000). *Fishing down the value chain: Modelling the impact of biodiversity loss in freshwater fisheries-the case of Malawi*. https://citeseerx.ist.psu.edu/document?repid=rep1&type=pdf&doi=569d9d3751dee6aea8016ee8d179e7f75eae36b7

Kateregga, E., & Sterner, T. (2009). Lake Victoria fish stocks and the effects of water hyacinth. *The Journal of Environment & Development*, *18*(1), 62–78. https://doi.org/10.1177/1070496508329467

Kathuria, P. C., & Rai, M. (2021). Case series of seven cases of urticaria, angioedema, and anaphylaxis (LTP syndrome) due to foods (nuts, lentils, and citrus foods) related to tree pollen (*Prosopis juliflora* and *Holoptelea integrifolia*) sensitization. *Indian Journal of Case Reports*, 319–323. https://doi.org/10.32677/ijcr.v7i8.2974

Katsanevakis, S., Wallentinus, I., Zenetos, A., Leppäkoski, E., Çinar, M. E., Oztürk, B., Grabowski, M., Golani, D., & Cardoso, A. C. (2014). Impacts of invasive alien marine species on ecosystem services and biodiversity: A pan-European review. *Aquatic Invasions*, *9*(4), 391–423. https://doi.org/10.3391/ai.2014.9.4.01

Kauffman, T. C., Martin, C. W., & Valentine, J. F. (2018). Hydrological alteration exacerbates the negative impacts of invasive Eurasian milfoil *Myriophyllum spicatum* by creating hypoxic conditions in a northern Gulf of Mexico estuary. *Marine Ecology Progress Series*, *592*, 97–108. https://doi.org/10.3354/meps12517

Kaufman, L. (1992). Catastrophic Change in Species-Rich Freshwater Ecosystems. *BioScience*, *42*(11), 846–858. https://doi.org/10.2307/1312084

Kauri Protection Governance Group. (2022). *National (*Phytophtora agathidicida*) Pest Management Plan proposal* (p. 77) [Proposal to meet requirements of Section 61 of the Biosecurity Act]. Biosecurity New Zealand, Minister for Primary Industries. https://www.mpi.govt.nz/biosecurity/long-term-biosecurity-management-programmes/protecting-kauri-from-disease/

Kay, B. K., & Peng, H. B. (1992). *Xenopus laevis: Practical Uses in Cell and Molecular Biology*. Elsevier Science. https://books.google.co.jp/books?id=hNAGVgKoWpsC

Keith, L. M., Hughes, R. F., Sugiyama, L. S., Heller, W. P., Bushe, B. C., & Friday, J. B. (2015). First Report of *Ceratocystis* Wilt on `Ōhi`a (*Metrosideros polymorpha*). *Plant Disease. 99(9): 1276*, *99*(9), 1276. https://doi.org/10.1094/PDIS-12-14-1293-PDN

Kellnreitner, F., Pockberger, M., Asmus, R., & Asmus, H. (2013). Feeding interactions between the introduced ctenophore *Mnemiopsis leidyi* and juvenile herring *Clupea harengus* in the Wadden Sea. *Biological Invasions*, *15*(4), 871–884. https://doi.org/10.1007/s10530-012-0336-4

Kelly, E. L. A., Cannon, A. L., & Smith, J. E. (2020). Environmental impacts and implications of tropical carrageenophyte seaweed farming. *Conservation Biology: The Journal of the Society for Conservation Biology*, *34*(2), 326–337. https://doi.org/10.1111/cobi.13462

Kelsch, A., Takahashi, Y., Dasgupta, R., Mader, A. D., Johnson, B. A., & Kumar, P. (2020). Invasive alien species and local communities in socio-ecological production landscapes and seascapes: A systematic review and analysis. *Environmental Science & Policy*, *112*, 275–281. https://doi.org/10.1016/j.envsci.2020.06.014

Kemp, S. F., DeShazo, R. D., Moffitt, J. E., Williams, D. F., & Buhner II, W. A. (2000). Expanding habitat of the imported fire ant (*Solenopsis invicta*): A public health concern. *Journal of Allergy and Clinical Immunology*, *105*(4), 683–691. https://doi.org/10.1067/mai.2000.105707

Kenis, M., Roques, A., Santini, A., & Liebhold, A. M. (2017). Impact of non-native invertebrates and pathogens on market forest tree resources. In *Impact of biological invasions on ecosystem services* (pp. 103–117). Springer. https://link.springer.com/chapter/10.1007/978-3-319-45121-3_7

Kent, R., & Dorward, A. (2015). Livelihood responses to *Lantana camara* invasion and biodiversity change in southern India: Application of an asset function framework. *Regional Environmental Change*, *15*(2), 353–364. https://doi.org/10.1007/s10113-014-0654-4

Kesner, D., & Kumschick, S. (2018). Gastropods alien to South Africa cause severe environmental harm in their global alien ranges across habitats. *Ecology and Evolution*, *8*(16), 8273–8285. https://doi.org/10.1002/ece3.4385

Kettenring, K. M., de Blois, S., & Hauber, D. P. (2012). Moving from a regional to a continental perspective of *Phragmites australis* invasion in North America. *AoB PLANTS*, *2012*. https://doi.org/10.1093/aobpla/pls040

Kettunen, M., Genovesi, P., Gollasch, S., Pagad, S., Starfinger, U., ten Brink, P., & Shine, C. (2009). *Technical support to EU strategy on invasive species (IAS)—Assessment of the impacts of IAS in Europe and the EU: Final report for the European Commission, Institute for European Environmental Policy (IEEP)* (pp. 44–44). Institute for European Environmental Policy (IEEP). https://researchspace.auckland.ac.nz/docs/uoa-docs/rights.htm

Kiełtyk, P., & Delimat, A. (2019). Impact of the alien plant *Impatiens glandulifera* on species diversity of invaded vegetation in the northern foothills of the Tatra Mountains, Central Europe. *Plant Ecology*, *220*(1), 1–12. https://doi.org/10.1007/s11258-018-0898-z

Killian, S., & McMichael, J. (2004). The human allergens of mesquite (*Prosopis juliflora*). *Clinical and Molecular Allergy*, *2*(1), 8. https://doi.org/10.1186/1476-7961-2-8

Kilpatrick, A. M. (2011). Globalization, Land Use, and the Invasion of West Nile Virus. *Science*, *334*(6054), 323–327. https://doi.org/10.1126/science.1201010

Kilpatrick, A. M., & Randolph, S. E. (2012). Drivers, dynamics, and control of emerging vector-borne zoonotic diseases. *Lancet*, *380*(9857), 1946–1955. https://doi.org/10.1016/S0140-6736(12)61151-9

Kindinger, T. L., & Albins, M. A. (2017). Consumptive and non-consumptive effects of an invasive marine predator on native coral-reef herbivores. *Biological Invasions*, *19*(1), 131–146. https://doi.org/10.1007/s10530-016-1268-1

Kinnear, J. E., Onus, M. L., & Bromilow, R. N. (1988). Fox control and rock-wallaby population dynamics. *Wildlife Research*, *15*(4), 435–450. https://www.publish.csiro.au/wr/pdf/wr9880435

Kinnear, J. E., Onus, M. L., & Sumner, N. R. (1998). Fox control and rock-wallaby population dynamics—II. An update. *Wildlife Research*, *25*(1), 81–88. https://www.publish.csiro.au/WR/WR96072

Kirichenko, N., Haubrock, P. J., Cuthbert, R. N., Akulov, E., Karimova, E., Shneider, Y., Liu, C., Angulo, E., Diagne, C., & Courchamp, F. (2021). Economic costs of biological invasions in terrestrial ecosystems in Russia. *NeoBiota*, *67*, 103–130. https://doi.org/10.3897/neobiota.67.58529

Klein, J. C., & Verlaque, M. (2009). Macrophyte assemblage associated with an invasive species exhibiting temporal variability in its development pattern. *Hydrobiologia*, *636*(1), 369–378. https://doi.org/10.1007/s10750-009-9966-7

Klein, J. C., & Verlaque, M. (2011). Experimental removal of the invasive *Caulerpa racemosa* triggers partial assemblage recovery. *Journal of the Marine Biological Association of the United Kingdom*, *91*(1), 117–125. https://doi.org/10.1017/S0025315410000792

Kleinschroth, F., Winton, R. S., Calamita, E., Niggemann, F., Botter, M., Wehrli, B., & Ghazoul, J. (2021). Living with floating vegetation invasions. *Ambio*, *50*(1), 125–137. https://doi.org/10.1007/s13280-020-01360-6

Klimaszyk, P., Klimaszyk, D., Piotrowiak, M., & Popiołek, A. (2014). Unusual complications after occupational exposure to giant hogweed (*Heracleum mantegazzianum*): A case report. *International Journal of Occupational Medicine and Environmental Health*, *27*(1), 141–144. https://doi.org/10.2478/s13382-014-0238-z

Knowler, D., & Barbier, E. (2005). Importing exotic plants and the risk of invasion: Are market-based instruments adequate? *Ecological Economics*, *52*(3), 341–354. https://doi.org/10.1016/j.ecolecon.2004.06.019

Kochalski, S., Riepe, C., Fujitani, M., Aas, Ø., & Arlinghaus, R. (2019). Public perception of river fish biodiversity in four European countries. *Conservation Biology*, *33*(1), 164–175. https://doi.org/10.1111/cobi.13180

Kochmann, J., Buschbaum, C., Volkenborn, N., & Reise, K. (2008). Shift from native mussels to alien oysters: Differential effects of ecosystem engineers. *Journal of Experimental Marine Biology and Ecology*, *364*(1), 1–10. https://doi.org/10.1016/j.jembe.2008.05.015

Koffi, D., Kyerematen, R., Eziah, V. Y., Osei-Mensah, Y. O., Afreh-Nuamah, K., Aboagye, E., Osae, M., & Meagher, R. L. (2020). Assessment of impacts of fall armyworm, *Spodoptera frugiperda* (Lepidoptera: Noctuidae) on maize production in Ghana. *Journal of Integrated Pest Management*, *11*(1), 20. https://doi.org/10.1093/jipm/pmaa015

Kohli, R. K., Batish, D. R., Singh, H. P., & Dogra, K. S. (2006). Status, invasiveness and environmental threats of three tropical American invasive weeds (*Parthenium hysterophorus* L., *Ageratum conyzoides* L., *Lantana camara* L.) in India. *Biological Invasions*, *8*(7), 1501–1510. https://doi.org/10.1007/s10530-005-5842-1

Koichi, K., Sangha, K. K., Cottrell, A., & Gordon, I. J. (2012). Aboriginal Rangers' Perspectives on Feral Pigs: Are they a Pest or a Resource? A Case Study in the Wet Tropics World Heritage Area of Northern Queensland. *Journal of Australian Indigenous Issues*, *15*(1), 2–20. https://www.researchgate.net/profile/Kamaljit-Sangha-2/publication/260752873_Aboriginal_Rangers'_Perspectives_on_Feral_Pigs_Are_they_a_Pest_or_a_Resource_A_Case_Study_in_the_Wet_Tropics_World_Heritage_Area_of_Northern_Queensland/links/00b7d53224cfdb852f000000/Aboriginal-Rangers-Perspectives-on-Feral-Pigs-Are-they-a-Pest-or-a-Resource-A-Case-Study-in-the-Wet-Tropics-World-Heritage-Area-of-Northern-Queensland.pdf

Kosoy, M., & Bai, Y. (2019). *Bartonella* bacteria in urban rats: A movement from the jungles of Southeast Asia to metropoles around the globe. *Frontiers in Ecology and Evolution*, *7*, 88. https://www.frontiersin.org/articles/10.3389/fevo.2019.00088/full

Kostecki, C., Rochette, S., Girardin, R., Blanchard, M., Desroy, N., & Le Pape, O. (2011). Reduction of flatfish habitat as a consequence of the proliferation of an invasive mollusc. *Estuarine, Coastal and Shelf Science*, *92*(1), 154–160. https://doi.org/10.1016/j.ecss.2010.12.026

Kotta, J., Kotta, I., Simm, M., Lankov, A., Lauringson, V., Põllumäe, A., & Ojaveer, H. (2006). Ecological consequences of biological invasions: Three invertebrate case studies in the north-eastern Baltic Sea. *Helgoland Marine Research*, *60*(2), 106. https://doi.org/10.1007/s10152-006-0027-6

Kouba, A., Oficialdegui, F. J., Cuthbert, R. N., Kourantidou, M., South, J., Tricarico, E., Leroy, B., Gozlan, R., Courchamp, F., & Haubrock, P. J. (2021). Feeling the pinch: Global economic costs of crayfish invasions and comparison with other aquatic crustaceans. *Science of the Total Environment*, *813*. https://doi.org/10.21203/rs.3.rs-381161/v1

Krishnan, M., & Kumar, R. N. (2010). *Socio-economic dimensions of seaweed farming in India* (p. 98). Central Marine Fisheries Research Institute. https://www.researchgate.net/profile/M-Krishnan/publication/262917111_Krishnan_M_and_R_Narayanakumar_2010_Socio-economic_dimensions_of_Seaweed_Farming_in_India_CMFRI_Special_Bulletin_No_102_p_78/links/02e7e539463102302f000000/Krishnan-M-and-R-Narayanakumar-2010-Socio-economic-dimensions-of-Seaweed-Farming-in-India-CMFRI-Special-Bulletin-No-102-p-78.pdf

Krueger, C. C., & May, B. (1991). Ecological and genetic effects of salmonid introductions in North America. *Canadian Journal of Fisheries and Aquatic Sciences*, *48*(S1), 66–77. https://doi.org/10.1139/f91-305

Kuebbing, S. E., & Nuñez, M. A. (2016). Invasive non-native plants have a greater effect on neighbouring natives than other non-natives. *Nature Plants*, *2*(10), 1–7. https://doi.org/10.1038/nplants.2016.134

Kull, C. A., Harimanana, S. L., Radaniela Andrianoro, A., & Rajoelison, L. G. (2019). Divergent perceptions of the 'neo-Australian' forests of lowland eastern Madagascar: Invasions, transitions, and livelihoods. *Journal of Environmental Management*, *229*, 48–56. https://doi.org/10.1016/j.jenvman.2018.06.004

Kull, C. A., Shackleton, C. M., Cunningham, P. J., Ducatillon, C., Dufour-Dror, J.-M., Esler, K. J., Friday, J. B., Gouveia, A. C., Griffin, A. R., Marchante, E., Midgley, S. J., Pauchard, A., Rangan, H., Richardson, D. M., Rinaudo, T., Tassin, J., Urgenson, L. S., von Maltitz, G. P., Zenni, R. D., & Zylstra, M. J. (2011). Adoption, use and perception of Australian acacias around the world. *Diversity and Distributions*, *17*(5), 822–836. https://doi.org/10.1111/j.1472-4642.2011.00783.x

Kumar, B. (2019). Rapid bioinvasion of alien mussel *Mytella strigata* (Hanley, 1843) (Bivalvia: Mytilidae) along Kerala coast, India: Will this impact the livelihood of fishers in Ashtamudi Lake? *Journal of Aquatic Biology & Fisheries*, *7*, 31–45. http://keralamarinelife.in/Journals/Vol7-12/5_Bijukumar_etal.pdf

Kumela, T., Simiyu, J., Sisay, B., Likhayo, P., Mendesil, E., Gohole, L., & Tefera, T. (2019). Farmers' knowledge, perceptions, and management practices of the new invasive pest, fall armyworm (*Spodoptera frugiperda*) in Ethiopia and Kenya. *International Journal of Pest Management*, *65*(1), 1–9. https://doi.org/10.1080/09670874.2017.1423129

Kumschick, S., Gaertner, M., Vilà, M., Essl, F., Jeschke, J. M., Pyšek, P., Ricciardi, A., Bacher, S., Blackburn, T. M., Dick, J. T. A., Evans, T., Hulme, P. E., Kühn, I., Mrugała, A., Pergl, J., Rabitsch, W., Richardson, D. M., Sendek, A., & Winter, M. (2015). Ecological Impacts of Alien Species: Quantification, Scope, Caveats, and Recommendations. *BioScience*, *65*(1), 55–63. https://doi.org/10.1093/biosci/biu193

Kwon, E., Ferguson, T. M., Park, S. Y., Manuzak, A., Qvarnstrom, Y., Morgan, S., Ciminera, P., & Murphy, G. S. (2013). A Severe Case of Angiostrongylus Eosinophilic Meningitis with Encephalitis and Neurologic Sequelae in Hawai'i. *Hawai'i Journal of Medicine & Public Health*, *72*(6 Suppl 2), 41–45. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC3689496/

La Marca, E., Lips, K. R., Lotters, S., Puschendorf, R., Ibanez, R., Rueda-Almonacid, J. V., Schulte, R., Marty, C., Castro, F., Manzanilla-Puppo, J., Garcia-Perez, J. E., Bolanos, F., Chaves, G., Pounds, J. A., Toral, E., & Young, B. E. (2005). Catastrophic Population Declines and Extinctions in Neotropical Harlequin Frogs (Bufonidae: *Atelopus*). *Biotropica*, *37*(2), 190–201. https://doi.org/10.1111/j.1744-7429.2005.00026.x

Lach, L. (2005). Interference and exploitation competition of three nectar-thieving invasive ant species. *Insectes Sociaux*, *52*(3), 257–262. https://doi.org/10.1007/s00040-005-0807-z

Lampo, M., Sánchez, D., Nicolás, A., Márquez, M., Nava-González, F., García, C. Z., Rinaldi, M., Rodríguez-Contreras, A., León, F., Han, B. A., & others. (2008). *Batrachochytrium dendrobatidis* in Venezuela. *Herpetological Review*, *39*(4), 449. https://www.researchgate.net/publication/262273041_Batrachochytrium_dendrobatidis_in_Venezuela

Lankau, R. A. (2012). Coevolution between invasive and native plants driven by chemical competition and soil biota. *Proceedings of the National Academy of Sciences*, *109*(28), 11240–11245. https://doi.org/10.1073/pnas.1201343109

Lankau, R. A., Nuzzo, V., Spyreas, G., & Davis, A. S. (2009). Evolutionary limits ameliorate the negative impact of an invasive plant. *Proceedings of the National Academy of Sciences*, *106*(36), 15362–15367. https://doi.org/10.1073/pnas.0905446106

LaPointe, D. A. (2000). *Avian malaria in Hawai'i: The distribution, ecology and vector potential of forest-dwelling mosquitoes* [University of Hawai'i at Manoa]. https://hilo.hawaii.edu/maunakea/library/reference.php?view=121

LaPointe, D. A. (2008). Dispersal of *Culex quinquefasciatus* (Diptera: Culicidae) in a Hawaiian rain forest. *Journal of Medical Entomology*, *45*(4), 600–609. https://doi.org/10.1093/jmedent/45.4.600

LaPointe, D. A. (2021). *Culex quinquefasciatus (southern house mosquito)*. CABI Compendium, CABI International. https://doi.org/10.1079/cabicompendium.86848

Laras, K., Sukri, N. C., Larasati, R. P., Bangs, M. J., Kosim, R., Djauzi, Wandra, T., Master, J., Kosasih, H., Hartati, S., Beckett, C., Sedyaningsih, E. R., Beecham, H. J., III, & Corwin, A. L. (2005). Tracking the re-emergence of epidemic chikungunya virus in Indonesia. *Transactions of The Royal Society of Tropical Medicine and Hygiene*, *99*(2), 128–141. https://doi.org/10.1016/j.trstmh.2004.03.013

Larson, S., Stoeckl, N., Fachry, M. E., Mustafa, M. D., Lapong, I., Purnomo, A. H., Rimmer, M. A., & Paul, N. A. (2021). Women's well-being and household benefits from seaweed farming in Indonesia. *Aquaculture*, *530*, 735711. https://doi.org/10.1016/j.aquaculture.2020.735711

Latombe, G., Pyšek, P., Jeschke, J. M., Blackburn, T. M., Bacher, S., Capinha, C., Costello, M. J., Fernández, M., Gregory, R. D., Hobern, D., Hui, C., Jetz, W., Kumschick, S., McGrannachan, C., Pergl, J., Roy, H. E., Scalera, R., Squires, Z. E., Wilson, J. R. U., … McGeoch, M. A. (2017). A vision for global monitoring of biological invasions. *Biological Conservation*, *213*, 295–308. https://doi.org/10.1016/j.biocon.2016.06.013

Lauber, T. B., Stedman, R. C., Connelly, N. A., Ready, R. C., Rudstam, L. G., & Poe, G. L. (2020). The effects of aquatic invasive species on recreational fishing participation and value in the Great Lakes: Possible future scenarios. *Journal of Great Lakes Research*, *46*(3), 656–665. https://doi.org/10.1016/j.jglr.2020.04.003

Lavretsky, P. (2020). Population genomics provides key insights into admixture, speciation, and evolution of closely related ducks of the mallard complex. In *Population Genomics: Wildlife* (pp. 295–330). Springer. https://link.springer.com/chapter/10.1007/13836_2020_76

Le Maitre, D. C., Gaertner, M., Marchante, E., Ens, E.-J., Holmes, P. M., Pauchard, A., O'Farrell, P. J., Rogers, A. M., Blanchard, R., Blignaut, J., & Richardson, D. M. (2011). Impacts of invasive Australian acacias: Implications for management and restoration: Australian acacias: linking impacts and restoration. *Diversity and Distributions*, *17*(5), 1015–1029. https://doi.org/10.1111/j.1472-4642.2011.00816.x

Le Pape, O., Guérault, D., & Désaunay, Y. (2004). Effect of an invasive mollusc, American slipper limpet *Crepidula fornicata*, on habitat suitability for juvenile common sole *Solea solea* in the Bay of Biscay. *Marine Ecology Progress Series*, *277*, 107–115. https://doi.org/10.3354/meps277107

Le Roux, J. J. (2021). *The Evolutionary Ecology of Invasive Species* (1st edition). Academic Press.

Le Roux, J. J., Clusella-Trullas, S., Mokotjomela, T. M., Mairal, M., Richardson, D. M., Skein, L., Wilson, J. R., Weyl, O. L., & Geerts, S. (2020). Biotic interactions as mediators of biological invasions: Insights from South Africa. In B. W. van Wilgen, J. Measy, D. M. Richardson, & T. A. Zengeya (Eds.), *Biological invasions in South Africa.* (pp. 387-427.). Springer. https://link.springer.com/chapter/10.1007/978-3-030-32394-3_14

Le Roux, V., Chapuis, J.-L., Frenot, Y., & Vernon, P. (2002). Diet of the house mouse (*Mus musculus*) on Guillou Island, Kerguelen archipelago, Subantarctic. *Polar Biology*, *25*(1), 49–57. https://doi.org/10.1007/s003000100310

Lebouvier, M., Laparie, M., Hullé, M., Marais, A., Cozic, Y., Lalouette, L., Vernon, P., Candresse, T., Frenot, Y., & Renault, D. (2011). The significance of the sub-Antarctic Kerguelen Islands for the assessment of the vulnerability of native communities to climate change, alien insect invasions and plant viruses. *Biological Invasions*, *13*, 1195–1208. https://doi.org/10.1007/s10530-011-9946-5

Leclerc, C., Courchamp, F., & Bellard, C. (2018). Insular threat associations within taxa worldwide. *Scientific Reports*, *8*(1), 6393. https://doi.org/10.1038/s41598-018-24733-0

Lehtiniemi, M., & Gorokhova, E. (2008). Predation of the introduced cladoceran *Cercopagis pengoi* on the native copepod *Eurytemora affinis* in the northern Baltic Sea. *Marine Ecology Progress Series*, *362*, 193–200.

Lengyel, N. (2009). The invasive colonial ascidian *Didemnum vexillum* on Georges Bank—Ecological effects and genetic identification. *Aquatic Invasions*, *4*(1), 143–152. https://doi.org/10.3391/ai.2009.4.1.15

Lenzner, B., Leclère, D., Franklin, O., Seebens, H., Roura-Pascual, N., Obersteiner, M., Dullinger, S., & Essl, F. (2019). A Framework for Global Twenty-First Century Scenarios and Models of Biological Invasions. *BioScience*, *69*(9), 697–710. https://doi.org/10.1093/biosci/biz070

León-Gamboa, A. L., Ramos, C., & García, M. R. (2010). Efecto de plantaciones de pino en la artropofauna del suelo de un bosque Altoandino. *Revista de Biología Tropical*, *58*(3), 1031–1048. https://www.scielo.sa.cr/scielo.php?pid=S0034-77442010000300016&script=sci_arttext

Leroy, B., Diagne, C., Angulo, E., Ballesteros-Mejia, L., Adamjy, T., Assailly, C., Albert, C., Andrews, L., Balzani, P., Banerjee, A. K., Bang, A., Bartlett, A., Bernery, C., Bodey, T., Bradshaw, C. J. A., Bufford, J., Capinha, C., Catford, J., Cuthbert, R., … Xiong, W. (2021). *Global Costs of Biological Invasions: Living Figure*. https://borisleroy.com/invacost/invacost_livingfigure.html

Leroy, B., Kramer, A. M., Vaissière, A., Kourantidou, M., Courchamp, F., & Diagne, C. (2022). Analysing economic costs of invasive alien species with the INVACOST R package. *Methods in Ecology and Evolution*, *13*(9), 1930–1937. https://doi.org/10.1111/2041-210X.13929

Lesser, M. P., & Slattery, M. (2011). Phase shift to algal dominated communities at mesophotic depths associated with lionfish (*Pterois volitans*) invasion on a Bahamian coral reef. *Biological Invasions*, *13*(8), 1855–1868. https://doi.org/10.1007/s10530-011-0005-z

Leung, B., Roura-Pascual, N., Bacher, S., Heikkila, J., Brotons, L., Burgman, M. A., Dehnen-Schmutz, K., Essl, F., Hulme, P. E., Richardson, D. M., Sol, D., & Vilà, M. (2012). TEASIng apart alien species risk assessments: A framework for best practices. *Ecology Letters*, *15*(12), 1475–1493. https://doi.org/10.1111/ele.12003

Levi, F., & Francour, P. (2004). Behavioural response of *Mullus surmuletus* to habitat modification by the invasive macroalga *Caulerpa taxifolia*. *Journal of Fish Biology*, *64*(1), 55–64. https://doi.org/10.1111/j.1095-8649.2004.00280.x

Lewin, W.-C., Arlinghaus, R., & Mehner, T. (2006). Documented and potential biological impacts of recreational fishing: Insights for management and conservation. *Reviews in Fisheries Science*, *14*(4), 305–367. https://doi.org/10.1080/10641260600886455

Leyse, K. E., Lawler, S. P., & Strange, T. (2004). Effects of an alien fish, *Gambusia affinis*, on an endemic California fairy shrimp, *Linderiella occidentalis*: Implications for conservation of diversity in fishless waters. *Biological Conservation*, *118*(1), 57–65. https://doi.org/10.1016/j.biocon.2003.07.008

Lim, S. K., Clements, J., & Khan, K. (2021). 793 The Giant Hogweed as A Rare Cause of Chemical Burns: A Case Series. *British Journal of Surgery*, *108*(Supplement_2). https://doi.org/10.1093/bjs/znab134.326

Limpus, C., & Reimer, D. (1994). The loggerhead turtle, *Caretta caretta*, in Queensland: A population in decline. In R. James (Ed.), *Proceedings of the Australian Marine Turtle Conservation Workshop* (pp. 39–47). Australian Nature Conservation Agency: Canberra.

Lindberg, C., Griffiths, C. L., & Anderson, R. J. (2020). Colonisation of South African kelp-bed canopies by the alien mussel *Mytilus galloprovincialis*: Extent and implications of a novel bioinvasion. *African Journal of Marine Science*, *42*(2), 167–176. https://doi.org/10.2989/1814232X.2020.1754908

Linders, T. E., Bekele, K., Schaffner, U., Allan, E., Alamirew, T., Choge, S. K., Haji, J., Muturi, G., Mbaabu, P. R., Shiferaw, H., & Eschen, R. (2020). The impact of invasive species on social-ecological systems: Relating supply and use of selected provisioning ecosystem services. *Ecosystem Services*, *41*, 101055. https://doi.org/10.1016/j.ecoser.2019.101055

Lindgren, C. J., Castro, K. L., Coiner, H. A., Nurse, R. E., & Darbyshire, S. J. (2013). The biology of invasive alien plants in Canada. 12. *Pueraria montana* var. *Lobata* (Willd.) Sanjappa & Predeep. *Canadian Journal of Plant Science*, *93*(1), 71–95. https://doi.org/10.4141/cjps2012-128

Linz, G. M., Homan, H. J., Gaulker, S. M., Penry, L. B., & Bleier, W. J. (2007). European starlings: A review of an invasive species with far-reaching impacts. *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, 24. https://digitalcommons.unl.edu/nwrcinvasive/24

Liu, C., Diagne, C., Angulo, E., Banerjee, A.-K., Chen, Y., Cuthbert, R. N., Haubrock, P. J., Kirichenko, N., Pattison, Z., Watari, Y., Xiong, W., & Courchamp, F. (2021).

Economic costs of biological invasions in Asia. *NeoBiota*, *67*, 53–78. https://doi.org/10.3897/neobiota.67.58147

Lockwood, J. L., Lieurance, D., Flory, S. L., Meyerson, L. A., Ricciardi, A., & Simberloff, D. (2023). Moving scholarship on invasion science forward. *Trends in Ecology & Evolution*, *38*(6). https://doi.org/10.1016/j.tree.2023.01.006

Lodge, D. M. (1993). Biological invasions: Lessons for ecology. *Trends in Ecology and Evolution*, *8*(4), Article 4. https://doi.org/10.1016/0169-5347(93)90025-K

Lodge, D. M., Deines, A., Gherardi, F., Yeo, D. C. J., Arcella, T., Baldridge, A. K., Barnes, M. A., Chadderton, W. L., Feder, J. L., Gantz, C. A., Howard, G. W., Jerde, C. L., Peters, B. W., Peters, J. A., Sargent, L. W., Turner, C. R., Wittmann, M. E., & Zeng, Y. (2012). Global introductions of crayfishes: Evaluating the impact of species invasions on ecosystem services. *Annual Review of Ecology, Evolution, and Systematics*, *43*(1), 449–472. https://doi.org/10.1146/annurev-ecolsys-111511-103919

Lojkić, I., Šimić, I., Bedeković, T., & Krešić, N. (2021). Current Status of Rabies and Its Eradication in Eastern and Southeastern Europe. *Pathogens*, *10*(6), 742. https://doi.org/10.3390/pathogens10060742

Lopez, B. E., Allen, J. M., Dukes, J. S., Lenoir, J., Vilà, M., Blumenthal, D. M., Beaury, E. M., Fusco, E. J., Laginhas, B. B., Morelli, T. L., O'Neill, M. W., Sorte, C. J. B., Maceda-Veiga, A., Whitlock, R., & Bradley, B. A. (2022). Global environmental changes more frequently offset than intensify detrimental effects of biological invasions. *Proceedings of the National Academy of Sciences*, *119*(22), e2117389119. https://doi.org/10.1073/pnas.2117389119

López Rosas, H., Moreno-Casasola, P., & Mendelssohn, I. A. (2005). Effects of an African grass invasion on vegetation, soil and interstitial water characteristics in a tropical freshwater marsh in La Mancha, Veracruz (Mexico). *Journal of Plant Interactions*, *1*(3), 187–195. https://doi.org/10.1080/17429140600857693

López-Farrán, Z., Guillaumot, C., Vargas-Chacoff, L., Paschke, K., Dulière, V., Danis, B., Poulin, E., Saucède, T., Waters, J., & Gérard, K. (2021). Is the southern crab *Halicarcinus planatus* (Fabricius, 1775) the next invader of Antarctica? *Global Change Biology*, *10*. https://doi.org/10.1111/gcb.15674

Lorenzo, P., Rodríguez-Echeverría, S., González, L., & Freitas, H. (2010). Effect of invasive *Acacia dealbata* Link on soil microorganisms as determined by PCR-DGGE. *Applied Soil Ecology*, *44*(3), 245–251. https://doi.org/10.1016/j.apsoil.2010.01.001

Lounibos, L. P. (2002). Invasions by insect vectors of human disease. *Annual Review of Entomology*, *47*, 233–266. https://doi.org/10.1146/annurev.ento.47.091201.145206

Lowe, S., Browne, M., Boudjelas, S., & De Poorter, M. (2000). *100 of the world's worst invasive alien species: A selection from the global invasive species database*. Invasive Species Specialist Group Auckland, New Zealand. www.issg.org/booklet.pdf

Ludyanskiy, M. L., McDonald, D., & MacNeill, D. (1993). Impact of the Zebra Mussel, a Bivalve Invader: *Dreissena polymorpha* is rapidly colonizing hard surfaces throughout waterways of the United States and Canada. *BioScience*, *43*(8), 533–544. https://doi.org/10.2307/1311948

Luizza, M. W., Wakie, T., Evangelista, P. H., & Jarnevich, C. S. (2016). Integrating local pastoral knowledge, participatory mapping, and species distribution modeling for risk assessment of invasive rubber vine (*Cryptostegia grandiflora*) in Ethiopia's Afar region. *Ecology and Society*, *21*(1). https://doi.org/10.5751/ES-07988-210122

Luomba, J. O. (2016). *Illegal, unreported and unregulated (IUU) fishing as a governability problem: A case study of Lake Victoria, Tanzania* [Masters, Memorial University of Newfoundland]. https://research.library.mun.ca/12424/

Lydeard, C., Cowie, R. H., Ponder, W. F., Bogan, A. E., Bouchet, P., Clark, S. A., Cummings, K. S., Frest, T. J., Gargominy, O., Herbert, D. G., Hershler, R., Perez, K. E., Roth, B., Seddon, M., Strong, E. E., & Thompson, F. G. (2004). The Global Decline of Nonmarine Mollusks. *BioScience*, *54*(4), 321–330. https://doi.org/10.1641/0006-3568(2004)054[0321:TGDONM]2.0.CO;2

Ma, K. C. K., Zardi, G. I., McQuaid, C. D., & Nicastro, K. R. (2020). Historical and contemporary range expansion of an invasive mussel, *Semimytlius algosus*, in Angola and Namibia despite data scarcity in an infrequently surveyed region. *PLoS ONE*, *15*(9), e0239167. https://doi.org/10.1371/journal.pone.0239167

Mabrouk, L., Ben Brahim, M., Jebara, A., & Jribi, I. (2021). Comparison of epiphyte algal assemblages on the leaves of marine seagrasses *Posidonia oceanica* (L.) Delile, *Cymodocea nodosa* (Ucria) Asch, and the lessepsian *Halophila stipulacea* (Forssk.) Asch in Chebba (East of Tunisia). *Marine Ecology*, *42*(2), e12642. https://doi.org/10.1111/maec.12642

Macdonald, I. A. W., Graber, D. M., DeBenedetti, S., Groves, R. H., & Fuentes, E. R. (1988). Introduced species in nature reserves in Mediterranean-type climatic regions of the world. *Biological Conservation*, *44*(1–2), 37–66. https://doi.org/10.1016/0006-3207(88)90004-3

Machekano, H., Mvumi, B., & Nyamukondiwa, C. (2017). Diamondback Moth, *Plutella xylostella* (L.) in Southern Africa: Research Trends, Challenges and Insights on Sustainable Management Options. *Sustainability*, *9*(2), 91. https://doi.org/10.3390/su9020091

Maclean, K., Robinson, C., Bock, E., & Rist, P. (2022). Reconciling risk and responsibility on Indigenous country: Bridging the boundaries to guide knowledge sharing for cross-cultural biosecurity risk management in northern Australia. *Journal of Cultural Geography*, *39*(1), 32–54. https://doi.org/10.1080/08873631.2021.1911078

MacLennan, C. (2017). Waves of migration: Settlement and creation of the Hawaiian environment. In *Environmental History of Modern Migrations*. Routledge. https://www.taylorfrancis.com/chapters/edit/10.4324/9781315731100-3/waves-migration-carol-maclennan

Macnaughton, A. E., Carvajal-Vallejos, F. M., Argote, A., Rainville, T. K., Van Damme, P. A., & Carolsfeld, J. (2015). "*Paiche* reigns!" species introduction and indigenous fisheries in the Bolivian Amazon. *Maritime Studies*, *14*(1), 11. https://doi.org/10.1186/s40152-015-0030-0

Magara, Y., Matsui, Y., Goto, Y., & Yuasa, A. (2001). Invasion of the non-indigenous nuisance mussel, *Limnoperna fortunei*, into water supply facilities in Japan. *Journal of Water Supply: Research and Technology-Aqua*, *50*(3), 113–124. https://doi.org/10.2166/aqua.2001.0011

Magnacca, K. N. (2007). *Conservation Status of the Endemic Bees of Hawai'i, Hylaeus* (Nesoprosopis) *(Hymenoptera: Colletidae)*. http://scholarspace.manoa.hawaii.edu/handle/10125/22606

Magnacca, K. N., & King, C. B. A. (2013). *Assessing the Presence and Distribution of 23 Hawaiian Yellow-Faced Bee Species on Lands Adjacent to Military Installations on O'ahu and Hawai'i Island* [Report]. Pacific Cooperative Studies Unit, University of Hawaii at Manoa. http://scholarspace.manoa.hawaii.edu/handle/10125/34064

Maldonado Andrade, G. (2019). *The Paradox of Culturally Useful Invasive Species: Chuspatel (Typha domingensis) Crafts of Lake Patzcuaro, Mexico* [M.A., California State University, Fullerton]. https://www.proquest.com/openview/57f112edbe28563f7b45504b68086835/1?pq-origsite=gscholar&cbl=18750&diss=y

Mallison, C. T., Stocker, R. K., & Cichra, C. E. (2001). Physical and vegetative characteristics of floating islands. *Journal of Aquatic Plant Management*, *39*, 107–111. https://www.researchgate.net/publication/237555790_Physical_and_Vegetative_Characteristics_of_Floating_Islands

Malvy, D., Ezzedine, K., Receveur, M.-C., Pistone, T., Crevon, L., Lemardeley, P., & Josse, R. (2008). Cluster of eosinophilic meningitis attributable to *Angiostrongylus cantonensis* infection in French policemen troop returning from the Pacific Islands. *Travel Medicine and Infectious Disease*, *6*(5), 301–304. https://doi.org/10.1016/j.tmaid.2008.06.003

Malyshev, A., & Quijón, P. A. (2011). Disruption of essential habitat by a coastal invader: New evidence of the effects of green crabs on eelgrass beds. *ICES Journal of Marine Science*, *68*(9), 1852–1856. https://doi.org/10.1093/icesjms/fsr126

Mank, J. E., Carlson, J. E., & Brittingham, M. C. (2004). A Century of Hybridization: Decreasing Genetic Distance Between American Black Ducks and Mallards. *Conservation Genetics*, *5*(3), 395–403. https://doi.org/10.1023/B:COGE.0000031139.55389.b1

Marais, C., Van Wilgen, B. W., & Stevens, D. (2004). The clearing of invasive alien plants in South Africa: A preliminary assessment of costs and progress. *South African Journal of Science*, *100*(1–2), 97–103.

Markert, A., Esser, W., Frank, D., Wehrmann, A., & Exo, K.-M. (2013). Habitat change by the formation of alien *Crassostrea*-reefs in the Wadden Sea and its role as feeding sites for waterbirds. *Estuarine, Coastal and Shelf Science*, *131*, 41–51. https://doi.org/10.1016/j.ecss.2013.08.003

Markert, A., Wehrmann, A., & Kröncke, I. (2010). Recently established *Crassostrea*-reefs *versus* native *Mytilus*-beds: Differences in ecosystem engineering affects the macrofaunal communities (Wadden Sea of Lower Saxony, southern German Bight). *Biological Invasions*, *12*(1), 15–32. https://doi.org/10.1007/s10530-009-9425-4

Marshall, B. E. (2018). Guilty as charged: Nile perch was the cause of the haplochromine decline in Lake Victoria. *Canadian Journal of Fisheries and Aquatic Sciences*, *75*(9), 1542–1559. https://doi.org/10.1139/cjfas-2017-0056

Martin, P., Dorn, N. J., Kawai, T., van der Heiden, C., & Scholtz, G. (2010). The enigmatic Marmorkrebs (marbled crayfish) is the parthenogenetic form of *Procambarus fallax* (Hagen, 1870). *Contributions to Zoology*, *79*(3), 107–118. https://doi.org/10.1163/18759866-07903003

Martínez, G. J., & Manzano-García, J. (2019). Perception and use of non-native and invasive flora from Sierras de Córdoba in central Argentina. *Acta Botanica Brasilica*, *33*(2), 241–253. https://doi.org/10.1590/0102-33062018abb0316

Martinez-Cillero, R., Willcock, S., Perez-Diaz, A., Joslin, E., Vergeer, P., & Peh, K. S. -H. (2019). A practical tool for assessing ecosystem services enhancement and degradation associated with invasive alien species. *Ecology and Evolution*, *9*(7), 3918–3936. https://doi.org/10.1002/ece3.5020

Martinou, A. F., & Roy, H. E. (2018). From local strategy to global frameworks: Effects of invasive non-native species on health and well-being. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health. CABI Invasive Species Series*. CABI. https://www.researchgate.net/publication/329124950_Introduction_From_Local_Strategy_to_Global_Frameworks_Effects_of_Invasive_Alien_Species_on_Health_and_Well-being

Martín-Torrijos, L., Kawai, T., Makkonen, J., Jussila, J., Kokko, H., & Diéguez-Uribeondo, J. (2018). Crayfish plague in Japan: A real threat to the endemic *Cambaroides japonicus*. *PLoS ONE*, *13*(4), e0195353. https://doi.org/10.1371/journal.pone.0195353

Marwoto, R. M., Heryanto, H., & Joshi, R. C. (2020). The invasive apple snail *Pomacea canaliculata* in Indonesia: A case study in Lake Rawa Pening, Central Java. *BIO Web of Conferences*, *19*, 00014. https://doi.org/10.1051/bioconf/20201900014

Matheson, K., McKenzie, C. H., Gregory, R. S., Robichaud, D. A., Bradbury, I. R., Snelgrove, P. V. R., & Rose, G. A. (2016). Linking eelgrass decline and impacts on associated fish communities to European green crab *Carcinus maenas* invasion. *Marine Ecology Progress Series*, *548*, 31–45. https://doi.org/10.3354/meps11674

Matsuzaki, S. S., Usio, N., Takamura, N., & Washitani, I. (2009). Contrasting impacts of invasive engineers on freshwater ecosystems: An experiment and meta-analysis. *Oecologia*, *158*(4), 673–686. https://doi.org/10.1007/s00442-008-1180-1

Maxwell, S. L., Fuller, R. A., Brooks, T. M., & Watson, J. E. M. (2016). Biodiversity: The ravages of guns, nets and bulldozers. *Nature News*, *536*(7615), 143. https://doi.org/10.1038/536143a

Mayee, C. D., Gujar, G. T., Dass, S., Balasubramanian, P., Kapoor, Y., & Choudhary, B. (2021). In retrospect: Managing an invasive pest, fall armyworm, *Spodoptera frugiperda*, in maize in India through digital and conventional networking pays off rich dividends towards crop sustainability. *Journal of Plant Diseases and Protection*. https://doi.org/10.1007/s41348-020-00411-0

McBeath, J. H., & McBeath, J. (2010). Invasive Species and Food Security. In *Environmental Change and Food Security in China* (Vol. 35, pp. 157–176). Springer. https://doi.org/10.1007/978-1-4020-9180-3_6

McCleery, R. A., Sovie, A., Reed, R. N., Cunningham, M. W., Hunter, M. E., & Hart, K. M. (2015). Marsh rabbit mortalities tie pythons to the precipitous decline of mammals in the Everglades. *Proceedings of the Royal Society B: Biological Sciences*, *282*(1805), 20150120. https://doi.org/10.1098/rspb.2015.0120

McClelland, G. T. W., Altwegg, R., van Aarde, R. J., Ferreira, S., Burger, A. E., & Chown, S. L. (2018). Climate change leads to increasing population density and impacts of a key island invader. *Ecological Applications*, *28*(1), 212–224. https://doi.org/10.1002/eap.1642

McGeoch, M. A., Shaw, J. D., Terauds, A., Lee, J. E., & Chown, S. L. (2015). Monitoring biological invasion across the broader Antarctic: A baseline and indicator framework. *Global Environmental Change*, *32*, 108–125. https://doi.org/10.1016/j.gloenvcha.2014.12.012

McKinnon, J. G., Gribben, P. E., Davis, A. R., Jolley, D. F., & Wright, J. T. (2009). Differences in soft-sediment macrobenthic assemblages invaded by *Caulerpa taxifolia* compared to uninvaded habitats. *Marine Ecology Progress Series*, *380*, 59–71. https://doi.org/10.3354/meps07926

McQuaid, K. A., & Griffiths, C. L. (2014). Alien reef-building polychaete drives long-term changes in invertebrate biomass and diversity in a small, urban estuary. *Estuarine, Coastal and Shelf Science*, *138*, 101–106. https://doi.org/10.1016/j.ecss.2013.12.016

Medina, F. M., Bonnaud, E., Vidal, E., Tershy, B. R., Zavaleta, E. S., Josh Donlan, C., Keitt, B. S., Corre, M., Horwath, S. V., & Nogales, M. (2011). A global review of the impacts of invasive cats on island endangered vertebrates. *Global Change Biology*, *17*(11), 3503–3510. https://doi.org/10.1111/j.1365-2486.2011.02464.x

Mejía, C., & Brandt, S. (2015). Managing tourism in the Galapagos Islands through price incentives: A choice experiment approach. *Ecological Economics*, *117*, 1–11. https://doi.org/10.1016/j.ecolecon.2015.05.014

Melotto, A., Manenti, R., & Ficetola, G. F. (2020). Rapid adaptation to invasive predators overwhelms natural gradients of intraspecific variation. *Nature Communications*, *11*(1), 3608. https://doi.org/10.1038/s41467-020-17406-y

Miguez Ruiz, A. P. (2013). *Problemática ecológica generada por el pez León Rojo (Pterois volitans) en la comunidad de la Isla de Providencia*. Pontificia Universidad Javeriana Facultad De Estudios Ambientales Y Rurales. https://repository.javeriana.edu.co/handle/10554/12493

Miller, A. E., Brosi, B. J., Magnacca, K., Daily, G. C., & Pejchar, L. (2015). Pollen Carried By Native and Nonnative Bees in the Large-scale Reforestation of Pastureland in Hawai 'i: Implications for Pollination. *Pacific Science*, *69*(1), 67–79. https://doi.org/10.2984/69.1.5

Mineur, F., Belsher, T., Johnson, M. P., Maggs, C. A., & Verlaque, M. (2007). Experimental assessment of oyster transfers as a vector for macroalgal introductions. *Biological Conservation*, *137*(2), 237–247. https://doi.org/10.1016/j.biocon.2007.02.001

Mineur, F., Le Roux, A., Maggs, C. A., & Verlaque, M. (2014). Positive feedback loop between introductions of non-native marine species and cultivation of oysters in Europe. *Conservation Biology: The Journal of the Society for Conservation Biology*, *28*(6), 1667–1676. https://doi.org/10.1111/cobi.12363

Miossec, L., Le Deuff, R.-M., & Goulletquer, P. (2009). Alien species alert: *Crassostrea gigas* (Pacific oyster). *ICES Cooperative Research Report*, *299*. https://archimer.ifremer.fr/doc/00000/6945/

Mirera, D. O., Kimathi, A., Ngarari, M. M., Magondu, E. W., Wainaina, M., & Ototo, A. (2020). Societal and environmental impacts of seaweed farming in relation to rural development: The case of Kibuyuni village, south coast, Kenya. *Ocean & Coastal Management*, *194*, 105253. https://doi.org/10.1016/j.ocecoaman.2020.105253

Miró, A., & Ventura, M. (2013). Historical use, fishing management and lake characteristics explain the presence of non-native trout in Pyrenean lakes: Implications for conservation. *Biological Conservation*, *167*, 17–24. https://doi.org/10.1016/j.biocon.2013.07.016

Modesto, V., Ilarri, M., Souza, A. T., Lopes-Lima, M., Douda, K., Clavero, M., & Sousa, R. (2018). Fish and mussels: Importance of fish for freshwater mussel conservation. *Fish and Fisheries*, *19*(2), 244–259. https://doi.org/10.1111/faf.12252

Molenaar, H., Meinesz, A., & Thibaut, T. (2009). Alterations of the structure of *Posidonia oceanica* beds due to the introduced alga *Caulerpa taxifolia*. *Scientia Marina*, *73*(2), 329–335. https://doi.org/10.3989/scimar.2009.73n2329

Molina-Montenegro, M. A., Bergstrom, D. M., Chwedorzewska, K. J., Convey, P., & Chown, S. L. (2019). Increasing impacts by Antarctica's most widespread invasive plant species as result of direct competition with native vascular plants. *NeoBiota*, *51*, 19–40. https://doi.org/10.3897/neobiota.51.37250

Möller, H., Spirén, A., Svensson, A., Gruvberger, B., Hindsén, M., & Bruze, M. (2002). Contact allergy to the Asteraceae plant *Ambrosia artemisiifolia* L (ragweed) in sesquiterpene lactone-sensitive patients in southern Sweden. *Contact Dermatitis*, *47*(3), 157–160. https://doi.org/10.1034/j.1600-0536.2002.470306.x

Molnár, K. (1993). Effect of decreased oxygen content on eels (*Anguilla anguilla*) infected by *Anguillicola crassus* (Nematoda: Dracunculoidea). *Acta Veterinaria Hungarica*, *41*(3–4), 349–360. https://pubmed.ncbi.nlm.nih.gov/8017238/

Molnár, K., Székely, C., & Baska, F. (1991). Mass mortality of eel in Lake Balaton due to *Anguillicola crassus* infection. *Bulletin of the European Association of Fish Pathologists*, *11*(6), 211–212. https://www.cabi.org/ISC/abstract/19940805353

Molyneux, D., Hallaj, Z., Keusch, G. T., McManus, D. P., Ngowi, H., Cleaveland, S., Ramos-Jimenez, P., Gotuzzo, E., Kar, K., & Sanchez, A. (2011). Zoonoses and marginalised infectious diseases of poverty: Where do we stand? *Parasites & Vectors*, *4*(1), 1–6. https://link.springer.com/article/10.1186/1756-3305-4-106

Monterroso, I., Binimelis, R., & Rodríguez-Labajos, B. (2011). New methods for the analysis of invasion processes: Multi-criteria evaluation of the invasion of *Hydrilla verticillata* in Guatemala. *Journal of Environmental Management*, *92*(3), 494–507. https://doi.org/10.1016/j.jenvman.2010.09.017

Mooney, H. A., & Drake, J. A. (1989). Biological invasions: A SCOPE program overview. In J. A. Drake, H. A. Mooney, R. H. Di Castri, F. J. Groves, & M. Kruger (Eds.), *Biological invasions: A global perspective.* (John Wiley, pp. 491–508).

Mori, E., Meini, S., Strubbe, D., Ancillotto, L., Sposimo, P., & Menchetti, M. (2018). Do alien free-ranging birds affect human health? A global summary of known zoonoses. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health*. CABI.

Morris, K., Johnson, B., Orell, P., Gaikhorst, G., Wayne, A., & Moro, D. (2003). Recovery of the threatened Chuditch (*Dasyurus geoffroii* Gould 1841): A case study. In M. Jones, C. Dickman, & M. Archer (Eds.), *'Predators with Pouches: The Biology of Carnivorous Marsupials'.* (pp. 435–451). CSIRO Publishing: Melbourne. https://www.researchgate.net/publication/49283463_Recovery_of_the_Threatened_Chuditch_Dasyurus_Geoffroll_A_Case_Study

Mosweu, S., Munyati, C., Kabanda, T., Setshogo, M., & Muzila, M. (2013). *Prosopis* L. Invasion in the South-Western Region of Botswana: The Perceptions of Rural Communities and Management Options. *Natural Resources*, *4*(8), Article 8. https://doi.org/10.4236/nr.2013.48061

Msuya, F. E., & Hurtado, A. Q. (2017). The role of women in seaweed aquaculture in the Western Indian Ocean and South-East Asia. *European Journal of Phycology*, *52*(4), 482–494. https://doi.org/10.1080/09670262.2017.1357084

Mukherjee, A., Velankar, A. D., & Kumara, H. N. (2017). Invasive *Prosopis juliflora* replacing the Native Floral Community over three decades: A case study of a World Heritage Site, Keoladeo National Park, India. *Biodiversity and Conservation*, *26*(12), 2839–2856. https://doi.org/10.1007/s10531-017-1392-y

Muller, G. C., Junnila, A., Traore, M. M., Traore, S. F., Doumbia, S., Sissoko, F., Dembele, S. M., Schlein, Y., Arheart, K. L., Revay, E. E., Kravchenko, V. D., Witt, A., & Beier, J. C. (2017). The invasive shrub *Prosopis juliflora* enhances the malaria parasite transmission capacity of *Anopheles* mosquitoes: A habitat manipulation experiment. *Malaria Journal*, *16*(1), 237. https://doi.org/10.1186/s12936-017-1878-9

Mungatana, E., & Ahimbisibwe, P. B. (2012). Qualitative impacts of *Senna spectabilis* on distribution of welfare: A household survey of dependent communities in Budongo Forest Reserve, Uganda. *Natural Resources Forum*, *36*(3), 181–191. https://doi.org/10.1111/j.1477-8947.2012.01454.x

Murray, K., Jepson, P. C., & Chaola, M. (2019). *Fall Armyworm Management for Maize Smallholders in Malawi: An Integrated Pest Management Strategic Plan*. CDMX, CIMMYT. https://repository.cimmyt.org/handle/10883/20170

Murray, K., Jepson, P. C., & Huesing, J. (2021). *Fall armyworm for maize smallholders in Kenya: An integrated pest management strategic plan*. USAID, CIMMYT. https://repository.cimmyt.org/handle/10883/21259

Muthukrishnan, R., Chiquillo, K. L., Cross, C., Fong, P., Kelley, T., Toline, C. A., Zweng, R., & Willette, D. A. (2020). Little giants: A rapidly invading seagrass alters ecosystem functioning relative to native foundation species. *Marine Biology*, *167*(81), 1–15. https://doi.org/10.1007/s00227-020-03689-8

Mwangi, E., & Swallow, B. (2008). *Prosopis juliflora* invasion and rural livelihoods in the Lake Baringo area of Kenya. *Conservation and Society*, *6*(2), 130–140. https://doi.org/10.4103/0972-4923.49207

Nash, D., Mostashari, F., Fine, A., Miller, J., O'Leary, D., Murray, K., Huang, A., Rosenberg, A., Greenberg, A., Sherman, M., Wong, S., Campbell, G. L., Roehrig, J. T., Gubler, D. J., Shieh, W.-J., Zaki, S., Smith, P., & Layton, M. (2001). The Outbreak of West Nile Virus Infection in the New York City Area in 1999. *New England Journal of Medicine*, *344*(24), 1807–1814. https://doi.org/10.1056/NEJM200106143442401

Neira, A., & Acero P, A. (2016). *Megalops atlanticus* (Megalopidae), un nuevo pez en el océano Pacífico; información sobre su importancia pesquera. *Revista MVZ Córdoba*, *21*(3), 5525–5534. https://doi.org/10.21897/rmvz.826

Nelufule, T., Robertson, M. P., Wilson, J. R. U., & Faulkner, K. T. (2022). Native-alien populations—An apparent oxymoron that requires specific conservation attention. *NeoBiota*, *74*, 57–74. https://doi.org/10.3897/neobiota.74.81671

Newbold, L. R., Hockley, F. A., Williams, C. F., Cable, J., Reading, A. J., Auchterlonie, N., & Kemp, P. S. (2015). Relationship between European eel *Anguilla anguilla* infection with non-native parasites and swimming behaviour on encountering accelerating flow. *Journal of Fish Biology*, *86*(5), 1519–1533. https://doi.org/10.1111/jfb.12659

Ngorima, A., & Shackleton, C. M. (2019). Livelihood benefits and costs from an invasive alien tree (*Acacia dealbata*) to rural communities in the Eastern Cape, South Africa. *Journal of Environmental Management*, *229*, 158–165. https://doi.org/10.1016/j.jenvman.2018.05.077

Nienhuis, C. M., Dietzsch, A. C., & Stout, J. C. (2009). The impacts of an invasive alien plant and its removal on native bees. *Apidologie*, *40*(4), 450–463. https://doi.org/10.1051/apido/2009005

Nonnis Marzano, F., Massimiliano, S., Chiesa, S., Gherardi, F., Piccinini, A., & Gibertini, G. (2009). The first record of the marbled crayfish adds further threats to fresh waters in Italy. *Aquatic Invasions*, *4*(2), 401–404. https://doi.org/10.3391/ai.2009.4.2.19

Norling, P., Lindegarth, M., Lindegarth, S., & Strand, Å. (2015). Effects of live and post-mortem shell structures of invasive Pacific oysters and native blue mussels on macrofauna and fish. *Marine Ecology Progress Series*, *518*, 123–138. https://doi.org/10.3354/meps11044

Norman, F. I. (1971). Predation by the fox (*Vulpes vulpes* L.) on colonies of the short-tailed shearwater (*Puffinus tenuirostris* (Temminck)) in Victoria, Australia. *Journal of Applied Ecology*, *8*(1), 21–32. https://doi.org/10.2307/2402124

Norwegian Biodiversity Information Centre. (2018). *Fremmedartslista [Alien species in Norway] 2018*. https://artsdatabanken.no/fab2018//N/537?mode=headless

Nuñez, M. A., & Amano, T. (2021). Monolingual searches can limit and bias results in global literature reviews. *Nature Ecology & Evolution*, *5*(3), 264. https://doi.org/10.1038/s41559-020-01369-w

Nuñez, M. A., Barlow, J., Cadotte, M., Lucas, K., Newton, E., Pettorelli, N., & Stephens, P. A. (2019). Assessing the uneven global distribution of readership, submissions and publications in applied ecology: Obvious problems without obvious solutions. *Journal of Applied Ecology*, *56*(1), 4–9. https://doi.org/10.1111/1365-2664.13319

Nuñez, M. A., Chiuffo, M. C., Seebens, H., Kuebbing, S., McCary, M. A., Lieurance, D., Zhang, B., Simberloff, D., & Meyerson, L. A. (2021). Two decades of data reveal that Biological Invasions needs to increase participation beyond North America, Europe, and Australasia. *Biological Invasions*, *24*, 333–340. https://doi.org/10.1007/s10530-021-02666-6

Nuñez, M. A., Pauchard, A., & Ricciardi, A. (2020). Invasion Science and the Global Spread of SARS-CoV-2. *Trends in Ecology & Evolution*, *35*(8), 642–645. https://doi.org/10.1016/j.tree.2020.05.004

Nuov, S., Viseth, H., & Vibol, O. (2005). Present status of alien species in aquaculture and aquatic ecosystem in Cambodia. *International Mechanisms for the Control and Responsible Use of Alien Species in Aquatic Ecosystems. Report of an Ad Hoc Expert Consultation.* 27-30 August 2003, Xishuangbanna, People's Republic of China. https://www.fao.org/3/a0113e/A0113E04.htm

Nurinsiyah, A. S., & Hausdorf, B. (2019). Listing, impact assessment and prioritization of introduced land snail and slug species in Indonesia. *Journal of Molluscan Studies*, *85*(1), 92–102. https://doi.org/10.1093/mollus/eyy062

Nyangau, P., Muriithi, B., Diiro, G., Akutse, K. S., & Subramanian, S. (2020). Farmers' knowledge and management practices of cereal, legume and vegetable insect pests, and willingness to pay for biopesticides. *International Journal of Pest Management*, *68*(3), 204–216. https://doi.org/10.1080/09670874.2020.1817621

Nyasembe, V. O., Cheseto, X., Kaplan, F., Foster, W. A., Teal, P. E. A., Tumlinson, J. H., Borgemeister, C., & Torto, B. (2015). The Invasive American Weed *Parthenium hysterophorus* Can Negatively Impact Malaria Control in Africa. *PLoS ONE*, *10*(9), e0137836. https://doi.org/10.1371/journal.pone.0137836

Obiri, J. F. (2011). Invasive plant species and their disaster-effects in dry tropical forests and rangelands of Kenya and Tanzania. *Jàmbá: Journal of Disaster Risk Studies*, *3*(2), 417–428. https://doi.org/10.4102/jamba.v3i2.39

O'Connor, T. G., & van Wilgen, B. W. (2020). The Impact of Invasive Alien Plants on Rangelands in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (pp. 459–487). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_16

Ogutu-Ohwayo, R. (1990). The decline of the native fishes of lakes Victoria and Kyoga

(East Africa) and the impact of introduced species, especially the Nile perch, *Lates niloticus*, and the Nile tilapia, *Oreochromis niloticus*. *Environmental Biology of Fishes*, *27*(2), 81–96. https://doi.org/10.1007/BF00001938

Ojaveer, H., Galil, B. S., Campbell, M. L., Carlton, J. T., Canning-Clode, J., Cook, E. J., Davidson, A. D., Hewitt, C. L., Jelmert, A., Marchini, A., McKenzie, C. H., Minchin, D., Occhipinti-Ambrogi, A., Olenin, S., & Ruiz, G. (2015). Classification of non-indigenous species based on their impacts: Considerations for application in marine management. *PLoS Biology*, *13*(4), e1002130. https://doi.org/10.1371/journal.pbio.1002130

Ojaveer, H., & Lumberg, A. (1995). On the Role of *Xerocopagis (Cercopagis) pengoi* (ostroumov) in Parnu Bay and the ne part Of the Gulf of Riga Ecosystem. *Proc. Esionian Acad. Sci. Ecol*, *5*(1/2), 20–25. https://www.academia.edu/download/44229342/On_the_role_of_Cercopagis_Ceropagis_pen20160330-22457-rvqauy.pdf

Ojaveer, H., Simm, M., & Lankov, A. (2004). Population dynamics and ecological impact of the non-indigenous *Cercopagis pengoi* in the Gulf of Riga (Baltic Sea). *Hydrobiologia*, *522*(1), 261–269. https://doi.org/10.1023/B:HYDR.0000029927.91756.41

Ólafsson, E., Johnstone, R. W., & Ndaro, S. G. M. (1995). Effects of intensive seaweed farming on the meiobenthos in a tropical lagoon. *Journal of Experimental Marine Biology and Ecology*, *191*(1), 101–117. https://doi.org/10.1016/0022-0981(95)00055-V

Olajuyigbe, O. O., & Afolayan, A. J. (2012). *In Vitro* Antibacterial and Time-Kill Assessment of Crude Methanolic Stem Bark Extract of *Acacia mearnsii* De Wild against Bacteria in Shigellosis. *Molecules*, *17*(2), 2103–2118. https://doi.org/10.3390/molecules17022103

Olden, J. D., Chen, K., García-Berthou, E., King, A. J., South, J., & Vitule, J. R. S. (2021). Invasive Species in Streams and Rivers. In *Reference Module in Earth Systems and Environmental Sciences* (Vol. 2, p. B9780128191668000839). Elsevier. https://doi.org/10.1016/B978-0-12-819166-8.00083-9

Olden, J. D., & Tamayo, M. (2014). Incentivizing the Public to Support Invasive Species Management: Eurasian Milfoil Reduces Lakefront Property Values. *PLoS ONE*, *9*(10), e110458. https://doi.org/10.1371/journal.pone.0110458

Olinger, L. K., Heidmann, S. L., Durdall, A. N., Howe, C., Ramseyer, T., Thomas, S. G., Lasseigne, D. N., Brown, E. J., Cassell, J. S., & Donihe, M. M. (2017). Altered juvenile fish communities associated with invasive *Halophila stipulacea* seagrass habitats in the US Virgin Islands. *PLoS One*, *12*(11), e0188386. https://doi.org/10.1371/journal.pone.0188386

Olson, S. L., & James, H. F. (1982). Fossil birds from the hawaiian islands: Evidence for wholesale extinction by man before Western contact. *Science*, *217*(4560), 633–635. https://doi.org/10.1126/science.217.4560.633

Olsson, M., Wapstra, E., Swan, G., Snaith, E. R. N., Clarke, R. O. N., & Madsen, T. (2005). Effects of long-term fox baiting on species composition and abundance in an Australian lizard community. *Austral Ecology*, *30*(8), 899–905. https://doi.org/10.1111/j.1442-9993.2005.01534.x

Ongore, C. O., Aura, C. M., Ogari, Z., Njiru, J. M., & Nyamweya, C. S. (2018). Spatial-temporal dynamics of water hyacinth, *Eichhornia crassipes* (Mart.) and other macrophytes and their impact on fisheries in Lake Victoria, Kenya. *Journal of Great Lakes Research*, *44*(6), 1273–1280. https://doi.org/10.1016/j.jglr.2018.10.001

Orizaola, G., & Brana, F. (2006). Effect of salmonid introduction and other environmental characteristics on amphibian distribution and abundance in mountain lakes of northern Spain. *Animal Conservation*, *9*(2), 171–178. https://doi.org/10.1111/j.1469-1795.2006.00023.x

Øyen, B.-H., & Nygaard, P. H. (2020). Impact of Sitka spruce on biodiversity in NW Europe with a special focus on Norway – evidence, perceptions and regulations. *Scandinavian Journal of Forest Research*, *35*(3–4), 117–133. https://doi.org/10.1080/02827581.2020.1748704

Öztürk, B., & İşinibilir, M. (2010). An alien jellyfish *Rhopilema nomadica* and its impacts to the Eastern Mediterranean part of Turkey. *J. Black Sea/Mediterranean Environment*, *16*(2), 10. https://blackmeditjournal.org/wp-content/uploads/2010-vol16-no2-1.pdf

Pagad, S., Genovesi, P., Carnevali, L., Schigel, D., & McGeoch, M. A. (2018). Introducing the Global Register of Introduced and Invasive Species. *Scientific Data*, *5*(1), 170202. https://doi.org/10.1038/sdata.2017.202

Page, L. K., Delzell, D. A. P., Gehrt, S. D., Harrell, E. D., Hiben, M., Walter, E., Anchor, C., & Kazacos, K. R. (2016). The Structure And Seasonality Of *Baylisascaris procyonis* Populations In Raccoons (*Procyon lotor*). *Journal of Wildlife Diseases*, *52*(2), 286–292. https://doi.org/10.7589/2015-06-153

Paini, D. R., Sheppard, A. W., Cook, D. C., De Barro, P. J., Worner, S. P., & Thomas, M. B. (2016). Global threat to agriculture from invasive species. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(27), 7575–7579. https://doi.org/10.1073/pnas.1602205113

Pallewatta, N., Reaser, J. K., & Gutierrez, A. T. (2003). *Invasive Alien Species in South-Southeast Asia*. 111. https://www.doi.gov/sites/doi.gov/files/uploads/invasive_alien_species_in_south_southeast_asia.pdf

Palstra, A. P., Heppener, D. F. M., van Ginneken, V. J. T., Székely, C., & van den Thillart, G. (2007). Swimming performance of silver eels is severely impaired by the swim-bladder parasite *Anguillicola crassus*. *Journal of Experimental Marine Biology and Ecology*, *352*(1), 244–256. https://core.ac.uk/outputs/16287669

Paritsis, J., Landesmann, J., Kitzberger, T., Tiribelli, F., Sasal, Y., Quintero, C., Dimarco, R., Barrios-García, M., Iglesias, A., Diez, J., Sarasola, M., & Nuñez, M. A. (2018). Pine Plantations and Invasion Alter Fuel Structure and Potential Fire Behavior in a Patagonian Forest-Steppe Ecotone. *Forests*, *9*(3), 117. https://doi.org/10.3390/f9030117

Parker, I. M., Simberloff, D., Lonsdale, W. M., Goodell, K., Wonham, M., Kareiva, P. M., Williamson, M. H., Von Holle, B., Moyle, P. B., Byers, J. E., & Goldwasser, L. (1999). Impact: Toward a Framework for Understanding the Ecological Effects of Invaders. *Biological Invasions*, *1*(1), 3–19. https://doi.org/10.1023/A:1010034312781

Pasiecznik, N. (2001). *The Prosopis juliflora—Prosopis pallida Complex: A Monograph*. Henry Doubleday Research Association (HDRA). https://www.gov.uk/research-for-development-outputs/the-prosopis-juliflora-prosopis-*pallida*-complex-a-monograph

Pasiecznik, N. (2022). *Prosopis juliflora* (mesquite). *CABI Compendium*, *CABI Compendium*, 43942. https://doi.org/10.1079/cabicompendium.43942

Patnaik, P., Abbasi, T., & Abbasi, S. A. (2017). *Prosopis* (*Prosopis juliflora*): Blessing and bane. *Tropical Ecology*, *58*(3), 455–483.

Patterson, E. J. K., & Bhatt, J. R. (2012). Impacts of cultivation of *Kappaphycus alvarezii* on coral reef environs of the Gulf of Mannar and Palk Bay, South-eastern India. *Invasive Alien Plants: An Ecological Appraisal for the Indian Subcontinent*, *1*, 89–98. https://books.google.co.jp/books?hl=ja&lr=lang_ja%7Clang_en&id=VzhKHiEojY4C&oi=fnd&pg=PA89&ots=uQIT8Sve7b&sig=2LekhM123e6Ly_H7oAZXl9lgUek&redir_esc=y#v=onepage&q&f=false

Patterson, E. J. K., Gilbert, M., Raj, K. D., Somesh Rajesh, A, A., & L, L. R. (2015). Invasion of the exotic seaweed, *Kappaphycus alvarezii*, on coral areas in two islands (Krusadai & Mulli) in the Gulf of Mannar, Southeastern India—Status and control measures. *Reef Encounter*, *30*(2), 51–53. https://www.researchgate.net/publication/323119080_Invasion_of_the_exotic_seaweed_Kappaphycus_alvarezii_on_coral_areas_in_two_islands_Krusadai_Mulli_in_the_Gulf_of_Mannar_Southeastern_India-status_and_control_measures

Pauchard, A., García, R. A., Peña, E., González, C., Cavieres, L. A., & Bustamante, R. O. (2008). Positive feedbacks between plant invasions and fire regimes: *Teline monspessulana* (L.) K. Koch (Fabaceae) in central Chile. *Biological Invasions*, *10*(4), 547–553. https://doi.org/10.1007/s10530-007-9151-8

Pauchard, A., García, R., Zalba, S., Sarasola, M., Zenni, R., Ziller, S., & Nuñez, M. A. (2015). 14. Pine Invasions in South America: Reducing Their Ecological Impacts Through Active Management. In J. Canning-Clode (Ed.), *Biological Invasions in Changing Ecosystems*. De Gruyter Open. https://doi.org/10.1515/9783110438666-020

Peacock, D. S., van Rensburg, B. J., & Robertson, M. P. (2007). The distribution and spread of the invasive alien common myna, *Acridotheres tristis* L.(Aves: Sturnidae), in southern Africa. *South African Journal of Science*, *103*(11–12), 465–473. http://www.scielo.org.za/scielo.php?script=sci_arttext&pid=S0038-23532007000600008

Pegado, C. M. A., Andrade, L. A. de, Félix, L. P., & Pereira, I. M. (2006). Efeitos da invasão biológica de algaroba: *Prosopis juliflora* (Sw.) DC. sobre a composição e a estrutura do estrato arbustivo-arbóreo da caatinga no Município de Monteiro, PB, Brasil. *Acta Botanica Brasilica*, *20*(4), 887–898. https://doi.org/10.1590/S0102-33062006000400013

Pejchar, L., & Mooney, H. A. (2009). Invasive species, ecosystem services and human well-being. *Trends in Ecology & Evolution*, *24*(9), 497–504. https://doi.org/10.1016/j.tree.2009.03.016

Perkins, R. C. L. (1913). *Introductory essay on the fauna* (Vol. 1). University Press.

Petran, A., & Moldoveanu, M. (1995). Post-invasion ecological impact of the Atlantic ctenophore *Mnemiopsis leidyi* Agassiz, 1865 on the zooplankton from the Romanian Black Sea waters. *Cercetari Marine*, *135*(137), 27–28. https://www.vliz.be/imisdocs/publications/ocrd/132739.pdf

Pfeiffer, J. M., & Voeks, R. A. (2008). Biological invasions and biocultural diversity: Linking ecological and cultural systems. *Environmental Conservation*, *35*(4), 281–293. https://doi.org/10.1017/S0376892908005146

Phillips, B. L., Brown, G. P., & Shine, R. (2003). Assessing the Potential Impact of Cane Toads on Australian Snakes. *Conservation Biology*, *17*(6), 1738–1747. https://doi.org/10.1111/j.1523-1739.2003.00353.x

Phillips, B. L., & Shine, R. (2004). Adapting to an invasive species: Toxic cane toads induce morphological change in Australian snakes. *Proceedings of the National Academy of Sciences*, *101*(49), 17150–17155. https://doi.org/10.1073/pnas.0406440101

Piazzi, L., Balata, D., & Cinelli, F. (2007). Invasions of alien macroalgae in Mediterranean coralligenous assemblages. *Cryptogamie Algologie*, *28*(3), 289–301. https://sciencepress.mnhn.fr/sites/default/files/articles/pdf/cryptogamie-algologie2008v28f3a17_1.pdf

Piazzi, L., Balata, D., Foresi, L., Cristaudo, C., & Cinelli, F. (2007). Sediment as a constituent of Mediterranean benthic communities dominated by *Caulerpa racemosa* var. Cylindracea. *Scientia Marina*, *71*(1), 129–135. https://doi.org/10.3989/scimar.2007.71n1129

Piazzi, L., & Ceccherelli, G. (2006). Persistence of biological invasion effects: Recovery of macroalgal assemblages after removal of *Caulerpa racemosa* var. *Cylindracea*. *Estuarine, Coastal and Shelf Science*, *68*(3–4), 455–461. https://doi.org/10.1016/j.ecss.2006.02.011

Piazzi, L., Ceccherelli, G., & Cinelli, F. (2001). Threat to macroalgal diversity: Effects of the introduced green alga *Caulerpa racemosa* in the Mediterranean. *Marine Ecology Progress Series*, *210*, 149–159. https://doi.org/10.3354/meps210149

Pimentel, D., Lach, L., Zuniga, R., & Morrison, D. (2000). Environmental and Economic Costs of Nonindigenous Species in the United States. *BioScience*, *50*(1), 53. https://doi.org/10.1641/0006-3568(2000)050[0053:EAECON]2.3.CO;2

Pimentel, D., McNair, S., Janecka, J., Wightman, J., Simmonds, C., O'Connell, C., Wong, E., Russel, L., Zern, J., Aquino, T., & Tsomondo, T. (2001). Economic and environmental threats of alien plant, animal, and microbe invasions. *Agriculture, Ecosystems & Environment*, *84*(1), Article 1. https://doi.org/10.1016/S0167-8809(00)00178-X

Pimentel, D., Zuniga, R., & Morrison, D. (2005). Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics*, *52*(3), 273–288. https://doi.org/10.1016/j.ecolecon.2004.10.002

Plentovich, S., Graham, J. R., Haines, W. P., & King, C. B. A. (2021). Invasive ants reduce nesting success of an endangered Hawaiian yellow-faced bee, *Hylaeus anthracinus*. *NeoBiota*, *64*, 137–154. https://doi.org/10.3897/neobiota.64.58670

Poirier, L. A., Symington, L. A., Davidson, J., St-Hilaire, S., & Quijón, P. A. (2017). Exploring the decline of oyster beds in Atlantic Canada shorelines: Potential effects of crab predation on American oysters (*Crassostrea virginica*). *Helgoland Marine Research*, *71*(1), 13. https://doi.org/10.1186/s10152-017-0493-z

Poland, T. M., Emery, M. R., Ciaramitaro, T., Pigeon, E., & Pigeon, A. (2017). Emerald ash borer, black ash, and Native american basketmaking. In E. Freedman & M. Meuzil, *Biodiversity, conservation, and environmental management in the Great Lakes basin.* (pp. 127–140). Routledge. https://www.taylorfrancis.com/chapters/edit/10.4324/9781315268774-11/emerald-ash-borer-black-ash-native-american-basketmaking-therese-poland-marla-emery-tina-ciaramitaro-ed-pigeon-angie-pigeon

Põllumäe, A., & Kotta, J. (2007). Factors describing the distribution of the zooplankton community in the Gulf of Finland in the context of interactions between native and introduced predatory cladocerans. *Oceanologia*, *49*(2), 277–290. http://www.iopan.gda.pl/oceanologia/

Polwiang, S. (2020). The time series seasonal patterns of dengue fever and associated weather variables in Bangkok (2003-2017). *BMC Infectious Diseases*, *20*(1), 208. https://doi.org/10.1186/s12879-020-4902-6

Potgieter, L. J., Gaertner, M., O'Farrell, P. J., & Richardson, D. M. (2019). Perceptions of impact: Invasive alien plants in the urban environment. *Journal of Environmental Management*, *229*(May 2018), 76–87. https://doi.org/10.1016/j.jenvman.2018.05.080

Pounds, J. A., Bustamante, M. R., Coloma, L. A., Consuegra, J. A., Fogden, M. P. L., Foster, P. N., La Marca, E., Masters, K. L., Merino-Viteri, A., Puschendorf, R., Ron, S. R., Sánchez-Azofeifa, G. A., Still, C. J., & Young, B. E. (2006). Widespread amphibian extinctions from epidemic disease driven by global warming. *Nature*, *439*(7073), 161–167. https://doi.org/10.1038/nature04246

Prahlow, J. A., & Barnard, J. J. (1998). Fatal anaphylaxis due to fire ant stings. *The American Journal of Forensic Medicine and Pathology*, *19*(2), 137–142. https://doi.org/10.1097/00000433-199806000-00007

Prasad, A. E. (2010). Effects of an exotic plant invasion on native understory plants in a tropical dry forest. *Conservation Biology*, *24*(3), 747–757. https://doi.org/10.1111/j.1523-1739.2009.01420.x

Pratt, C. F., Constantine, K. L., & Murphy, S. T. (2017). Economic impacts of invasive alien species on African smallholder livelihoods. *Global Food Security*, *14*, 31–37. https://doi.org/10.1016/j.gfs.2017.01.011

Pratt, T. K., Woodworth, B. L., Camp, R. J., & Gorresen, P. M. (2006). Hawaii Forest Bird Interagency Database Project: Collecting, Understanding, and Sharing Population Data on Hawaiian Forest Birds. In *Hawaii Forest Bird Interagency Database Project: Collecting, Understanding, and Sharing Population Data on Hawaiian Forest Birds* (USGS Numbered Series 2006–3013; Fact Sheet, Vols. 2006–3013, p. 2). Geological Survey (U.S.). https://doi.org/10.3133/fs20063013

Prebble, M., & Wilmshurst, J. M. (2009). Detecting the initial impact of humans and introduced species on island environments in Remote Oceania using palaeoecology. *Biological Invasions*, *11*(7), 1529–1556. https://doi.org/10.1007/s10530-008-9405-0

Pretty Paint-Small, V. (2013). *Linking culture, ecology and policy: The invasion of Russian-olive (*Elaeagnus angustifolia *l.) on the Crow Indian Reservation, south-central Montana, USA* [Thesis, Colorado State University. Libraries]. https://mountainscholar.org/handle/10217/78865

Puri, R. K. (2015). The uniqueness of the everyday: Herders and invasive species in India. In J. Barnes & M. R. Dove (Eds.), *Climate Cultures: Anthropological Perspectives on Climate Change* (pp. 249–272). Yale University Press. http://yalebooks.com/book/9780300198812/climate-cultures

Purvis, A., Molnár, Z., Obura, D., Ichii, K., Willis, K., Chettri, N., Dulloo, M., Hendry, A., Gabrielyan, B., Gutt, J., Jacob, U., Keskin, E., Niamir, A., Öztürk, B., Salimov, R., & Jaureguiberry, P. (2019). *Chapter 2.2 Status and Trends –Nature*. Zenodo. https://doi.org/10.5281/ZENODO.5041234

Pye, T., & Bonner, W. N. (1980). Feral Brown rats, *Rattus norvegicus*, in South Georgia (South Atlantic Ocean). *Journal of Zoology*, *192*(2), 237–255. https://doi.org/10.1111/j.1469-7998.1980.tb04232.x

Radford, J. Q., Woinarski, J. C. Z., Legge, S., Baseler, M., Bentley, J., Burbidge, A. A., Bode, M., Copley, P., Dexter, N., Dickman, C. R., Gillespie, G., Hill, B., Johnson, C. N., Kanowski, J., Latch, P., Letnic, M., Manning, A., Menkhorst, P., Mitchell, N., … Ringma, J. (2018). Degrees of population-level susceptibility of Australian terrestrial non-volant mammal species to predation by the introduced red fox (*Vulpes vulpes*) and feral cat (*Felis catus*). *Wildlife Research*, *45*(7), 645. https://doi.org/10.1071/WR18008

Raffaele, E., Nuñez, M. A., Eneström, J., & Blackhall, M. (2016). Fire as mediator of pine invasion: Evidence from Patagonia, Argentina. *Biological Invasions*, *18*(3), 597–601. https://doi.org/10.1007/s10530-015-1038-5

Raghubanshi, A. S., & Tripathi, A. (2009). Effect of disturbance, habitat fragmentation and alien invasive plants on floral diversity in dry tropical forests of Vindhyan highland: A review. *Tropical Ecology*, *50*(1), 57–69. https://www.cabdirect.org/cabdirect/abstract/20093131363

Rai, R. K., & Scarborough, H. (2015). Understanding the Effects of the Invasive Plants on Rural Forest-dependent Communities. *Small-Scale Forestry*, *14*(1), 59–72. https://doi.org/10.1007/s11842-014-9273-7

Rakauskas, V., Šidagytė-Copilas, E., Stakėnas, S., & Garbaras, A. (2020). Invasive *Neogobius melanostomus* in the Lithuanian Baltic Sea coast: Trophic role and impact on the diet of piscivorous fish. *Journal of Great Lakes Research*, *46*(3), 597–608. https://doi.org/10.1016/j.jglr.2020.03.005

Rameshkumar, S., & Rajaram, R. (2017). Experimental cultivation of invasive seaweed *Kappaphycus alvarezii* (Doty) Doty with assessment of macro and meiobenthos diversity from Tuticorin coast, Southeast coast of India. *Regional Studies in Marine Science*, *9*, 117–125. https://doi.org/10.1016/j.rsma.2016.12.002

Rameshkumar, S., & Rajaram, R. (2019). Impact of seaweed farming on socio-economic development of a fishing community in Palk Bay, Southeast Coast of India. In *Coastal Zone Management* (pp. 501–513). Elsevier. https://www.sciencedirect.com/science/article/abs/pii/B9780128143506000227

Rascher, K. G., Große-Stoltenberg, A., Máguas, C., Meira-Neto, J. A. A., & Werner, C. (2011). *Acacia longifolia* invasion impacts vegetation structure and regeneration dynamics in open dunes and pine forests. *Biological Invasions*, *13*(5), 1099–1113. https://doi.org/10.1007/s10530-011-9949-2

Rawson, P. D., Agrawal, V., & Hilbish, T. J. (1999). Hybridization between the blue mussels *Mytilus galloprovincialis* and *M. trossulus* along the Pacific coast of North America: Evidence for limited introgression. *Marine Biology*, *134*(1), 201–211. https://doi.org/10.1007/s002270050538

Read, J., Schlesinger, C., Firn, J., Ryan-Colton, E., Grice, T., & Murphy, R. (2020). Ranking buffel: Comparative risk and mitigation costs of key environmental and socio-cultural threats in central Australia. *Ecology and Evolution*, *10*(23), 12745–12763. https://doi.org/10.1002/ece3.6724

Régnier, C., Fontaine, B., & Bouchet, P. (2009). Not Knowing, Not Recording, Not Listing: Numerous Unnoticed Mollusk Extinctions. *Conservation Biology*, *23*(5), 1214–1221. https://doi.org/10.1111/j.1523-1739.2009.01245.x

Remedios-De León, M., Hughes, K. A., Morelli, E., & Convey, P. (2021). International response under the Antarctic Treaty System to the establishment of a non-native fly in Antarctica. *Environmental Management*, *67*(6), 1043–1059. https://link.springer.com/article/10.1007/s00267-021-01464-z

Renault, D., Angulo, E., Cuthbert, R., Haubrock, P. J., Capinha, C., Kramer, A. M., & Courchamp, F. (2022). The magnitude, diversity, and distribution of the economic costs of invasive terrestrial invertebrates worldwide. *Science of The Total Environment*, *835*. https://doi.org/10.1016/j.scitotenv.2022.155391

Renault, D., Manfrini, E., Leroy, B., Diagne, C., Ballesteros-Mejia, L., Angulo, E., & Courchamp, F. (2021). Biological invasions in France: Alarming costs and even more alarming knowledge gaps. *NeoBiota*, *67*, 191–224. https://doi.org/10.3897/neobiota.67.59134

Renčo, M., Kornobis, F. W., Domaradzki, K., Jakubska-Busse, A., Jurová, J., & Homolová, Z. (2019). How does an invasive *Heracleum sosnowskyi* affect soil nematode communities in natural conditions? *Nematology*, *21*(1), 71–89. https://doi.org/10.1163/15685411-00003196

Reo, N. J., Whyte, K., Ranco, D., Brandt, J., Blackmer, E., & Elliott, B. (2017). Invasive Species, Indigenous Stewards, and Vulnerability Discourse. *American Indian Quarterly*, *41*(3), 201–223. https://doi.org/10.5250/amerindiquar.41.3.0201

Rettberg, S. (2010). Contested narratives of pastoral vulnerability and risk in Ethiopia's Afar region. *Pastoralism – Research, Policy and Practice*, *1*(2), 248–273. https://www.humanitarianlibrary.org/sites/default/files/2013/07/s7.pdf

Reynolds, S. A., & Aldridge, D. C. (2021). Embracing the Allelopathic Potential of Invasive Aquatic Plants to Manipulate Freshwater Ecosystems. *Frontiers in Environmental Science*, *8*, 551803. https://www.frontiersin.org/articles/10.3389/fenvs.2020.551803/full

Rezza, G., Nicoletti, L., Angelini, R., Romi, R., Finarelli, A. C., Panning, M., Cordioli, P., Fortuna, C., Boros, S., Magurano, F., Silvi, G., Angelini, P., Dottori, M., Ciufolini, M. G., Majori, G. C., Cassone, A., & CHIKV study group. (2007). Infection with chikungunya virus in Italy: An outbreak in a temperate region. *Lancet (London, England)*, *370*(9602), 1840–1846. https://doi.org/10.1016/S0140-6736(07)61779-6

Rhoades, R. B., Stafford, C. T., & James, F. K. (1989). Survey of fatal anaphylactic reactions to imported fire ant stings. Report of the Fire Ant Subcommittee of the American Academy of Allergy and Immunology. *The Journal of Allergy and Clinical Immunology*, *84*(2), 159–162. https://doi.org/10.1016/0091-6749(89)90319-9

Rhymer, J. M., & Simberloff, D. (1996). Extinction By Hybridization and Introgression. *Annual Review of Ecology and Systematics*, *27*(1), 83–109. https://doi.org/10.1146/annurev.ecolsys.27.1.83

Rhymer, J. M., Williams, M. J., & Braun, M. J. (1994). Mitochondrial Analysis of Gene Flow between New Zealand Mallards (*Anas Platyrhynchos*) and Grey Ducks (*A. Superciliosa*). *The Auk*, *111*(4), Article 4. https://doi.org/10.2307/4088829

Ricciardi, A., Hoopes, M. F., Marchetti, M. P., & Lockwood, J. L. (2013). Progress toward understanding the ecological impacts of nonnative species. *Ecological Monographs*, *83*(3), 263–282. https://doi.org/10.1890/13-0183.1

Ricciardi, A., & Macisaac, H. J. (2011). Impacts of Biological Invasions on Freshwater Ecosystems. In D. M. Richardson (Ed.), *Fifty Years of Invasion Ecology* (pp. 211–224). Wiley-Blackwell. https://doi.org/10.1002/9781444329988.ch16

Ricciardi, A., Neves, R. J., & Rasmussen, J. B. (1998). Impending Extinctions of North American Freshwater Mussels (Unionoida) Following the Zebra Mussel (*Dreissena polymorpha*) Invasion. *Journal of Animal Ecology*, *67*(4), 613–619. https://www.jstor.org/stable/2647282

Richardson, D. M. (1998). Forestry Trees as Invasive Aliens. *Conservation Biology*, *12*(1), 18–26. https://doi.org/10.1046/j.1523-1739.1998.96392.x

Richardson, D. M., & Rejmanek, M. (2004). Conifers as invasive aliens: A global survey and predictive framework. *Diversity and Distributions*, *10*(5–6), Article 5–6. https://doi.org/10.1111/j.1366-9516.2004.00096.x

Rico-Sánchez, A. E., Haubrock, P. J., Cuthbert, R. N., Angulo, E., Ballesteros-Mejia, L., López-López, E., Duboscq-Carra, V. G., Nuñez, M. A., Diagne, C., & Courchamp, F. (2021). Economic costs of invasive alien species in Mexico. *NeoBiota*, *67*, 459–483. https://doi.org/10.3897/neobiota.67.63846

Riedmiller, S. (1994). Lake Victoria fisheries: The Kenyan reality and environmental implications. *Environmental Biology of Fishes*, *39*(4), 329–338. https://doi.org/10.1007/BF00004802

Riisgård, H. U., Bøttiger, L., Madsen, C. V., & Purcell, J. E. (2007). Invasive ctenophore *Mnemiopsis leidyi* in Limfjorden (Denmark) in late summer 2007-assessment of abundance and predation effects. *Aquatic Invasions*, *2*(4), 395–401. https://pdfs.semanticscholar.org/9a52/d558c5066d1f78e4083d753b0d99f75a55b6.pdf

Riisgård, H. U., Madsen, C. V., Barth-Jensen, C., & Purcell, J. E. (2012). Population dynamics and zooplankton-predation impact of the indigenous scyphozoan *Aurelia aurita* and the invasive ctenophore *Mnemiopsis leidyi* in Limfjorden (Denmark). *Aquatic Invasions*, *7*(2). http://dx.doi.org/10.3391/ai.2012.7.2.001

Rijal, S., & Cochard, R. (2016). Invasion of *Mimosa pigra* on the cultivated Mekong River floodplains near Kratie, Cambodia: Farmers' coping strategies, perceptions, and outlooks. *Regional Environmental Change*, *16*(3), 681–693. https://doi.org/10.1007/s10113-015-0776-3

Rimmer, M. A., Larson, S., Lapong, I., Purnomo, A. H., Pong-Masak, P. R., Swanepoel, L., & Paul, N. A. (2021). Seaweed aquaculture in Indonesia contributes to social and economic aspects of livelihoods and community wellbeing. *Sustainability*, *13*(19), 10946. https://www.mdpi.com/2071-1050/13/19/10946

Roberts, J., Chick, A., Oswald, L., & Thompson, P. (1995). Effect of carp, *Cyprinus carpio* L., an exotic benthivorous fish, on aquatic plants and water quality in experimental ponds. *Marine and Freshwater Research*, *46*(8), 1171–1180. https://www.publish.csiro.au/mf/MF9951171

Robeyns, I. (2005). The Capability Approach: A theoretical survey. *Journal of Human Development*, *6*(1), 93–117. https://doi.org/10.1080/146498805200034266

Robin, L., Robin, K., Camerlenghi, E., Ireland, L., & Ryan-Colton, E. (2022). How Dreaming and Indigenous ancestral stories are central to nature conservation: Perspectives from Walalkara Indigenous Protected Area, Australia. *Ecological Management & Restoration*, *23*(S1), 43–52. https://doi.org/10.1111/emr.12528

Robinson, C. J., Smyth, D., & Whitehead, P. J. (2005). Bush Tucker, Bush Pets, and Bush Threats: Cooperative Management of Feral Animals in Australia's Kakadu National Park. *Conservation Biology*, *19*(5), 1385–1391. https://doi.org/10.1111/j.1523-1739.2005.00196.x

Robinson, C. J., & Wallington, T. J. (2012). Boundary work: Engaging knowledge systems in co-management. *Ecology and Society*, *17*(2), 16. https://doi.org/10.5751/es-04836-170216

Robinson, T. B., Branch, G., Griffiths, C., Govender, A., & Hockey, P. (2007). Changes in South African rocky intertidal invertebrate community structure associated with the invasion of the mussel *Mytilus galloprovincialis*. *Marine Ecology Progress Series*, *340*, 163–171. https://doi.org/10.3354/meps340163

Robinson, T. B., & Griffiths, C. L. (2002). Invasion of Langebaan Lagoon, South

Africa, by *Mytilus galloprovincialis*–effects on natural communities. *African Zoology*, *37*(2), 151–158. https://doi.org/10.1080/15627020.2002.11657170

Rochford, M., Krysko, K. L., Nifong, J., Wilkins, L., Snow, R. W., & Cherkiss, M. S. (2010). *Python molurus bivittatus* (Burmese python). Diet. *Herpetol Rev*, *41*, 97. https://www.dropbox.com/s/vcoxlqlg2i2hzrf/HR%2BMarch%2B2010%2Bebook.pdf?dl=1

Rockwell-Postel, M., Laginhas, B. B., & Bradley, B. A. (2020). Supporting proactive management in the context of climate change: Prioritizing range-shifting invasive plants based on impact. *Biological Invasions*. https://doi.org/10.1007/s10530-020-02261-1

Roder, W., Phengchanh, S., Keoboualapha, B., & Maniphone, S. (1995). *Chromolaena odorata* in slash-and-burn rice systems of Northern Laos. *Agroforestry Systems*, *31*(1), 79–92. https://doi.org/10.1007/BF00712056

Rogers, P., Nunan, F., & Fentie, A. A. (2017). Reimagining invasions: The social and cultural impacts of *Prosopis* on pastoralists in southern Afar, Ethiopia. *Pastoralism*, *7*(1), 22. https://doi.org/10.1186/s13570-017-0094-0

Romi, R., Boccolini, D., Luca, M. di, Medlock, J. M., Schaffner, F., Severini, F., & Toma, L. (2018). The invasive mosquitoes of medical importance. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health* (Vol. 10, pp. 76–90). CABI. https://doi.org/10.1079/9781786390981.0076

Roy, H. E., Hesketh, H., Purse, B. V., Eilenberg, J., Santini, A., Scalera, R., Stentiford, G. D., Adriaens, T., Bacela-Spychalska, K., Bass, D., Beckmann, K. M., Bessell, P., Bojko, J., Booy, O., Cardoso, A. C., Essl, F., Groom, Q., Harrower, C., Kleespies, R., … Dunn, A. M. (2017). Alien Pathogens on the Horizon: Opportunities for Predicting their Threat to Wildlife. *Conservation Letters*, *10*(4), 477–484. https://doi.org/10.1111/conl.12297

Roy, K., Jaenecke, K. A., & Peck, R. W. (2020). Ambrosia Beetle (Coleoptera: Curculionidae) Communities and Frass Production in ʻŌhiʻa (Myrtales: Myrtaceae) Infected With *Ceratocystis* (Microascales: Ceratocystidaceae) Fungi Responsible for Rapid ʻŌhiʻa Death. *Environmental Entomology*, *49*(6), 1345–1354. https://doi.org/10.1093/ee/nvaa108

Ruiz, G. M., Fofonoff, P., Hines, A. H., & Grosholz, E. D. (1999). Non-indigenous species as stressors in estuarine and marine communities: Assessing invasion impacts and interactions. *Limnology and Oceanography*, *44*(3part2), 950–972. https://doi.org/10.4319/lo.1999.44.3_part_2.0950

Rupp, H. (1996). Adverse assessments of *Gambusia affinis*: An alternate view for mosquito control practitioners. *Journal of the American Mosquito Control Association*, *12*(2 Pt 1), 155–159; discussion 160-6. PubMed. http://europepmc.org/abstract/MED/8827587

Russell, D. J. (1983). *Ecology of the imported red seaweed* Eucheuma striatum *Schmitz on Coconut Island, Oahu, Hawaii*. http://hdl.handle.net/10125/650

Russell, J. C., Peace, J. E., Houghton, M. J., Bury, S. J., & Bodey, T. W. (2020). Systematic prey preference by introduced mice exhausts the ecosystem on Antipodes Island. *Biological Invasions*, *22*, 1265–1278. https://doi.org/10.1007/s10530-019-02194-4

Russell, S., Ens, E., & Ngukurr Yangbala Rangers. (2020). Connection as Country: Relational values of billabongs in Indigenous northern Australia. *Ecosystem Services*, *45*, 101169. https://doi.org/10.1016/j.ecoser.2020.101169

Russell, S., Ens, E., & Ngukurr Yangbala Rangers. (2021). "Now it's not a billabong": Eco-cultural assessment of billabong condition in remote northern Australia. *Marine and Freshwater Research*. https://doi.org/10.1071/MF20080

Rwomushana, I., Bateman, M., Beale, T., Beseh, P., Cameron, K., Chiluba, M., Clottey, V., Davis, T., Day, R., Early, R., Godwin, J., Gonzalez-Moreno, P., Kansiime, M., Kenis, M., Makale, F., Mugambi, I., Murphy, S., Nunda, W., Phiri, N., … Tambo, J. (2018). *Fall armyworm: Impacts and implications for Africa. Evidence Note Update, October 2018*. CABI. https://www.invasive-species.org/wp-content/uploads/sites/2/2019/02/FAW-Evidence-Note-October-2018.pdf

Sadchatheeswaran, S., Branch, G. M., & Robinson, T. B. (2015). Changes in habitat complexity resulting from sequential invasions of a rocky shore: Implications for community structure. *Biological Invasions*, *17*(6), 1799–1816. https://doi.org/10.1007/s10530-014-0837-4

Sadek, S., El-Soud, W., & Galil, B. (2018). The brown shrimp *Penaeus aztecus* Ives, 1891 (Crustacea, Decapoda, Penaeidae) in the Nile Delta, Egypt: An exploitable resource for fishery and mariculture? *BioInvasions Records*, *7*(1), 51–54. https://doi.org/10.3391/bir.2018.7.1.07

Sahli, H. F., Krushelnycky, P. D., Drake, D. R., & Taylor, A. D. (2016). Patterns of Floral Visitation to Native Hawaiian Plants in Presence and Absence of Invasive Argentine Ants. *Pacific Science*, *70*(3), 309–322. https://doi.org/10.2984/70.3.3

Salafsky, N., Salzer, D., Stattersfield, A. J., Hilton-Taylor, C., Neugarten, R., Butchart, S. H. M., Collen, B., Cox, N., Master, L. L., O'connor, S., & Wilkie, D. (2008). A Standard Lexicon for Biodiversity Conservation: Unified Classifications of Threats and Actions. *Conservation Biology*, *22*(4), 897–911. https://doi.org/10.1111/j.1523-1739.2008.00937.x

Salmón, E. (2000). Kincentric Ecology: Indigenous Perceptions of the Human-Nature Relationship. *Ecological Applications*, *10*(5), 1327–1332. https://doi.org/10.2307/2641288

Samuel, M. D., Woodworth, B. L., Atkinson, C. T., Hart, P. J., & LaPointe, D. A. (2015). Avian malaria in Hawaiian forest birds: Infection and population impacts across species and elevations. *Ecosphere*, *6*(6), art104. https://doi.org/10.1890/ES14-00393.1

Sánchez, J. A., & Ballesteros, D. (2014). The invasive snowflake coral (*Carijoa riisei*) in the Tropical Eastern Pacific, Colombia. *Revista de Biología Tropical*, *62*, 199. https://doi.org/10.15517/rbt.v62i0.16276

Sandvik, H., Hilmo, O., Henriksen, S., Elven, R., Åsen, P. A., Hegre, H., Pedersen, O., Pedersen, P. A., Solstad, H., Vandvik, V., Westergaard, K. B., Ødegaard, F., Åström, S., Elven, H., Endrestøl, A., Gammelmo, Ø., Hatteland, B. A., Solheim, H., Nordén, B., … Gederaas, L. (2020). Alien species in Norway: Results from quantitative ecological impact assessments. *Ecological Solutions and Evidence*, *1*(1), e12006. https://doi.org/10.1002/2688-8319.12006

Sangha, K. K., Le Brocque, A., Costanza, R., & Cadet-James, Y. (2015). Ecosystems and indigenous well-being: An integrated framework. *Global Ecology and Conservation*, *4*, 197–206. https://doi.org/10.1016/j.gecco.2015.06.008

Sangha, K. K., Preece, L., Villarreal-Rosas, J., Kegamba, J. J., Paudyal, K., Warmenhoven, T., & RamaKrishnan, P. S. (2018). An ecosystem services framework to evaluate indigenous and local peoples' connections with nature. *Ecosystem Services*, *31*, 111–125. https://doi.org/10.1016/j.ecoser.2018.03.017

Santini, A., Ghelardini, L., De Pace, C., Desprez-Loustau, M. L., Capretti, P., Chandelier, A., Cech, T., Chira, D., Diamandis, S., Gaitniekis, T., Hantula, J., Holdenrieder, O., Jankovsky, L., Jung, T., Jurc, D., Kirisits, T., Kunca, A., Lygis, V., Malecka, M., … Stenlid, J. (2013). Biogeographical patterns and determinants of invasion by forest pathogens in Europe. *New Phytologist*, *197*(1), 238–250. https://doi.org/10.1111/j.1469-8137.2012.04364.x

Santos, C. A. B., & Nóbrega Alves, R. R. (2016). Ethnoichthyology of the indigenous Truká people, Northeast Brazil. *Journal of Ethnobiology and Ethnomedicine*, *12*(1), 1. https://doi.org/10.1186/s13002-015-0076-5

Sapsford, S. J., Brandt, A. J., Davis, K. T., Peralta, G., Dickie, I. A., Gibson, R. D., Green, J. L., Hulme, P. E., Nuñez, M. A., Orwin, K. H., Pauchard, A., Wardle, D. A., & Peltzer, D. A. (2020). Towards a framework for understanding the context dependence of impacts of non-native tree species. *Functional Ecology*, *34*(5), 944–955. https://doi.org/10.1111/1365-2435.13544

Sato, T. (2013). Beyond water-intensive agriculture: Expansion of *Prosopis juliflora* and its growing economic use in Tamil Nadu, India. *Land Use Policy*, *35*, 283–292. https://doi.org/10.1016/j.landusepol.2013.06.001

Saul, W.-C., Jeschke, J., & Heger, T. (2013). The role of eco-evolutionary experience in invasion success. *NeoBiota*, *17*, 57–74. https://doi.org/10.3897/neobiota.17.5208

Saul, W.-C., & Jeschke, J. M. (2015). Eco-evolutionary experience in novel species interactions. *Ecology Letters*, *18*(3), 236–245. https://doi.org/10.1111/ele.12408

Saunders, G., Cooke, B., McColl, K., Shine, R., & Peacock, T. (2010). Modern approaches for the biological control of vertebrate pests: An Australian perspective. *Biological Control*, *52*(3), 288–295. https://doi.org/10.1016/j.biocontrol.2009.06.014

Saure, H. I., Vandvik, V., Hassel, K., & Vetaas, O. R. (2013). Effects of invasion by introduced *versus* native conifers on coastal heathland vegetation. *Journal of Vegetation Science*, *24*(4), 744–754. https://doi.org/10.1111/jvs.12010

Saure, H. I., Vandvik, V., Hassel, K., & Vetaas, O. R. (2014). Do vascular plants and bryophytes respond differently to coniferous invasion of coastal heathlands? *Biological Invasions*, *16*(4), Article 4. https://doi.org/10.1007/s10530-013-0536-6

Savary, S., Willocquet, L., Pethybridge, S. J., Esker, P., McRoberts, N., & Nelson, A. (2019). The global burden of pathogens and pests on major food crops. *Nature Ecology & Evolution*, *3*(3), 430–439. https://doi.org/10.1038/s41559-018-0793-y

Sayol, F., Steinbauer, M. J., Blackburn, T. M., Antonelli, A., & Faurby, S. (2020). Anthropogenic extinctions conceal widespread evolution of flightlessness in birds. *Science Advances*, *6*(49), eabb6095. https://doi.org/10.1126/sciadv.abb6095

Schaber, M., Haslob, H., Huwer, B., Harjes, A., Hinrichsen, H.-H., Storr-Paulsen, M., Schmidt, J. O., Voss, R., Neumann, V., & Köster, F. W. (2011). Spatio-temporal overlap of the alien invasive ctenophore *Mnemiopsis leidyi* and ichthyoplankton in the Bornholm Basin (Baltic Sea). *Biological Invasions*, *13*(12), 2647–2660. https://doi.org/10.1007/s10530-011-9936-7

Scheibling, R. E., Patriquin, D. G., & Filbee-Dexter, K. (2018). Distribution and abundance of the invasive seagrass *Halophila stipulacea* and associated benthic macrofauna in Carriacou, Grenadines, Eastern Caribbean. *Aquatic Botany*, *144*, 1–8. https://doi.org/10.1016/j.aquabot.2017.10.003

Schindler, S., Rabitsch, W., & Essl, F. (2018). Climate change and increase of impacts on human health by alien species. In G. Mazza & E. Tricarico (Eds.), *Invasive species and human health* (pp. 151–166). CABI. https://doi.org/10.1079/9781786390981.0151

Schloegel, L. M., Hero, J.-M., Berger, L., Speare, R., McDonald, K., & Daszak, P. (2006). The decline of the sharp-snouted day frog (*Taudactylus acutirostris*): The first documented case of extinction by infection in a free-ranging wildlife species? *EcoHealth*, *3*(1), 35–40. https://doi.org/10.1007/s10393-005-0012-6

Schneider, K., van der Werf, W., Cendoya, M., Mourits, M., Navas-Cortés, J. A., Vicent, A., & Oude Lansink, A. (2020). Impact of *Xylella fastidiosa* subspecies *pauca* in European olives. *Proceedings of the National Academy of Sciences*, *117*(17), 9250–9259. https://doi.org/10.1073/pnas.1912206117

Schwarz, D., Matta, B. M., Shakir-Botteri, N. L., & McPheron, B. A. (2005). Host shift to an invasive plant triggers rapid animal hybrid speciation. *Nature*, *436*(7050), 546–549. https://doi.org/10.1038/nature03800

Scott, J. J., & Kirkpatrick, J. B. (2008). Rabbits, landslips and vegetation change on the coastal slopes of subantarctic Macquarie Island, 1980–2007: Implications for management. *Polar Biology*, *31*(4), 409–419.

Sebastián, C. R., Steffani, C. N., & Branch, G. M. (2002). Homing and movement patterns of a South African limpet *Scutellastra argenvillei* in an area invaded by an alien mussel *Mytilus galloprovincialis*. *Marine Ecology Progress Series*, *243*, 111–122. http://dx.doi.org/10.3354/meps243111

Seebens, H., Bacher, S., Blackburn, T. M., Capinha, C., Dawson, W., Dullinger, S., Genovesi, P., Hulme, P. E., Kleunen, M., Kühn, I., Jeschke, J. M., Lenzner, B., Liebhold, A. M., Pattison, Z., Pergl, J., Pyšek, P., Winter, M., & Essl, F. (2021). Projecting the continental accumulation of alien species through to 2050. *Global Change Biology*, *27*(5), 970–982. https://doi.org/10.1111/gcb.15333

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., … Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications*, *8*(1), 14435. https://doi.org/10.1038/ncomms14435

Segev, O., Mangel, M., & Blaustein, L. (2009). Deleterious effects by mosquitofish (*Gambusia affinis*) on the endangered fire salamander (*Salamandra infraimmaculata*). *Animal Conservation*, *12*(1), 29–37. https://doi.org/10.1111/j.1469-1795.2008.00217.x

Sellers, A., Saltonstall, K., & Davidson, T. (2015). The introduced alga *Kappaphycus alvarezii* (Doty ex P.C. Silva, 1996) in abandoned cultivation sites in Bocas del Toro, Panama. *BioInvasions Records*, *4*(1), 1–7. https://doi.org/10.3391/bir.2015.4.1.01

Sen, A. (1999). *Commodities and Capabilities*. Oxford University Press.

Seo, K. S., & Lee, Y. (2009). A First Assessment of Invasive Marine Species on Chinese and Korean Coasts. In G. Rilov & J. A. Crooks (Eds.), *Biological Invasions in Marine Ecosystems* (Vol. 204, pp. 577–585). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-540-79236-9_32

Sghaier, Y. R., Zakhama-Sraieb, R., & Charfi-Cheikhrouha, F. (2014). Effects of the invasive seagrass *Halophila stipulacea* on the native seagrass *Cymodocea nodosa*. *Proceedings of the*

5th Mediterranean Symposium on Marine Vegetation, 167–171. http://dx.doi.org/10.13140/2.1.3086.6888

Shackleton, C. M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S. E., & Fabricius, C. (2007). Assessing the Effects of Invasive Alien Species on Rural Livelihoods: Case Examples and a Framework from South Africa. *Human Ecology*, *35*(1), 113–127. https://doi.org/10.1007/s10745-006-9095-0

Shackleton, R. T., Foxcroft, L. C., Pyšek, P., Wood, L. E., & Richardson, D. M. (2020). Assessing biological invasions in protected areas after 30 years: Revisiting nature reserves targeted by the 1980s SCOPE programme. *Biological Conservation*, *243*, 108424. https://doi.org/10.1016/j.biocon.2020.108424

Shackleton, R. T., Le Maitre, D. C., Pasiecznik, N. M., & Richardson, D. M. (2014). *Prosopis*: A global assessment of the biogeography, benefits, impacts and management of one of the world's worst woody invasive plant taxa. *AoB PLANTS*, *6*, plu027. https://doi.org/10.1093/aobpla/plu027

Shackleton, R. T., Richardson, D. M., Shackleton, C. M., Bennett, B., Crowley, S. L., Dehnen-Schmutz, K., Estévez, R. A., Fischer, A., Kueffer, C., Kull, C. A., Marchante, E., Novoa, A., Potgieter, L. J., Vaas, J., Vaz, A. S., & Larson, B. M. H. (2019). Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management*, *229*, 10–26. https://doi.org/10.1016/j.jenvman.2018.04.045

Shackleton, R. T., Shackleton, C. M., & Kull, C. A. (2019). The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management*, *229*, 145–157. https://doi.org/10.1016/j.jenvman.2018.05.007

Shackleton, R. T., Witt, A. B. R., Piroris, F. M., & van Wilgen, B. W. (2017). Distribution and socio-ecological impacts of the invasive alien cactus *Opuntia stricta* in eastern Africa. *Biological Invasions*, *19*(8), 2427–2441. https://doi.org/10.1007/s10530-017-1453-x

Shackleton, S. E., & Shackleton, R. T. (2018). Local knowledge regarding ecosystem services and disservices from invasive alien plants in the arid Kalahari, South Africa. *Journal of Arid Environments*, *159*, 22–33. https://doi.org/10.1016/j.jaridenv.2017.07.001

Sharma, G. P. (2011). *Lantana camara* L. invasion and impacton herb layer diversity and soil properties in a dry deciduous forest of India. *Applied Ecology and Environmental Research*, *9*(3), 253–264. https://doi.org/10.15666/aeer/0903_253264

Shields, J. l., Heath, J., & Heath, D. D. (2010). Marine landscape shapes hybrid zone in a broadcast spawning bivalve: Introgression and genetic structure in Canadian west coast *Mytilus*. *Marine Ecology Progress Series*, *399*, 211–223. https://doi.org/10.3354/meps08338

Shiferaw, H., Alamirew, T., Dzikiti, S., Bewket, W., Zeleke, G., & Schaffner, U. (2021). Water use of *Prosopis juliflora* and its impacts on catchment water budget and rural livelihoods in Afar Region, Ethiopia. *Scientific Reports*, *11*(1), 2688. https://doi.org/10.1038/s41598-021-81776-6

Shiferaw, H., Bewket, W., Alamirew, T., Zeleke, G., Teketay, D., Bekele, K., Schaffner, U., & Eckert, S. (2019). Implications of land use/land cover dynamics and *Prosopis* invasion on ecosystem service values in Afar Region, Ethiopia. *Science of The Total Environment*, *675*, 354–366. https://doi.org/10.1016/j.scitotenv.2019.04.220

Shiferaw, H., Schaffner, U., Bewket, W., Alamirew, T., Zeleke, G., Teketay, D., & Eckert, S. (2019). Modelling the current fractional cover of an invasive alien plant and drivers of its invasion in a dryland ecosystem. *Scientific Reports*, *9*(1), 1576. https://doi.org/10.1038/s41598-018-36587-7

Shiganova, T. A. (1997). *Mnemiopsis leidyi* abundance in the Black Sea and its impact on the pelagic community. In *Sensitivity to Change: Black Sea, Baltic Sea and North Sea* (pp. 117–129). Springer. https://link.springer.com/chapter/10.1007/978-94-011-5758-2_10

Shiganova, T. A. (1998). Invasion of the Black Sea by the ctenophore *Mnemiopsis leidyi* and recent changes in pelagic community structure. *Fisheries Oceanography*, *7*(3–4), Article 3–4. https://doi.org/10.1046/j.1365-2419.1998.00080.x

Shiganova, T. A., & Bulgakova, Y. V. (2000). Effects of gelatinous plankton on Black Sea and Sea of Azov fish and their food resources. *ICES Journal of Marine Science*, *57*(3), 641–648. https://doi.org/10.1006/jmsc.2000.0736

Shiganova, T. A., Musaeva, E. I., Bulgakova, Y. V., Mirzoyan, Z. A., & Martynyuk, M. L. (2003). Invaders ctenophores *Mnemiopsis leidyi* (A. Agassiz) and *Beroe ovata* Mayer 1912, and their influence on the pelagic ecosystem of Northeastern Black Sea. *Biology Bulletin of the Russian Academy of Sciences*, *30*(2), 180–190. https://link.springer.com/article/10.1023/A:1023249508158

Simberloff, D. (2010). Charles Elton: Neither Founder Nor Siren, but Prophet. In *Fifty Years of Invasion Ecology* (pp. 11–24). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781444329988.ch2

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vilà, M. (2013). Impacts of biological invasions: What's what and the way forward. *Trends in Ecology & Evolution*, *28*(1), 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Simberloff, D., & Von Holle, B. (1999). Positive Interactions of Nonindigenous Species: Invasional Meltdown? *Biological Invasions*, *1*(1), 21–32. https://doi.org/10.1023/A:1010086329619

Singer, M. C., & Parmesan, C. (2018). Lethal trap created by adaptive evolutionary response to an exotic resource. *Nature*, *557*(7704), 238–241. https://doi.org/10.1038/s41586-018-0074-6

Singh, A. K., & Lakra, W. S. (2011). Risk and benefit assessment of alien fish species of the aquaculture and aquarium trade into India: Risks and benefits of alien species in India. *Reviews in Aquaculture*, *3*(1), 3–18. https://doi.org/10.1111/j.1753-5131.2010.01039.x

Singh, A. K., Pathak, A. K., & Lakra, W. S. (2010). Invasion of an Exotic Fish—Common Carp, *Cyprinus Carpio* L. (Actinopterygii: Cypriniformes: Cyprinidae) in the Ganga River, India and its Impacts. *Acta Ichthyologica Et Piscatoria*, *40*(1), 11–19. https://doi.org/10.3750/AIP2010.40.1.02

Singh, N., Shukla, S., Gupta, V., Tandia, N., & Singh, P. (2015). Mosquito borne zoonotic diseases: A review. *Journal of Livestock Science*, *6*, 65–72. http://livestockscience.in/wp-content/uploads/singh-zoonotic-disease.pdf

Sinha, B. C., Goyal, S. P., & Krauseman, P. R. (2009). How the use of mesquite impacts grass availability, Wild Ass Sanctuary, India. *Desert Plants*, *25*(2), 3–9. https://www.cabdirect.org/cabdirect/abstract/20103010299

Sjöberg, N. B., Petersson, E., Wickström, H., & Hansson, S. (2009). Effects of the swimbladder parasite *Anguillicola crassus* on the migration of European silver eels *Anguilla anguilla* in the Baltic Sea. *Journal of Fish Biology*, *74*(9), 2158–2170. https://doi.org/10.1111/j.1095-8649.2009.02296.x

Skein, L., Alexander, M., & Robinson, T. (2018). Contrasting invasion patterns in intertidal and subtidal mussel communities. *African Zoology*, *53*(1), 47–52. https://doi.org/10.1080/15627020.2018.1448720

Skurikhina, L. A., Kartavtsev, Y. F., Chichvarkhin, A. Y., & Pan'kova, M. V. (2001). Study of Two Species of Mussels, *Mytilus trossulusand Mytilus galloprovincialis* (Bivalvia, Mytilidae), and Their Hybrids in Peter the Great Bay of the Sea of Japan with the Use of PCR Markers. *Russian Journal of Genetics*, *37*(12), 1448–1451. https://doi.org/10.1023/A:1013264400526

Sloane, D. R., Ens, E., Wunungmurra, J., Falk, A., Marika, G., Maymuru, M., Towler, G., Preece, D., & the Yirralka Rangers. (2019). Western and Indigenous knowledge converge to explain *Melaleuca* forest dieback on Aboriginal land in northern Australia. *Marine and Freshwater Research*, *70*(1), 125. https://doi.org/10.1071/MF18009

Sloane, D. R., Ens, E., Wunungmurra, Y., Gumana, Y., Wunungmurra, B., Wirrpanda, M., Towler, G., Preece, D., & Rangers, Y. (2021). Lessons from old fenced plots: Eco-cultural Impacts of feral ungulates and potential decline in sea-level rise resilience of coastal floodplains in northern Australia. *Ecological Management & Restoration*, *22*(2), 191–203. https://doi.org/10.1111/emr.12464

Smith, K. G. (2020). *The IUCN Red List and invasive alien species: An analysis of impacts on threatened species and extinctions*. IUCN. http://www.issg.org/pdf/publications/IUCN-Red-List-IAS.pdf

Smith, K., Grundy, J., & Nelson, H. J. (2010). Culture at the centre of community based aged care in a remote Australian Indigenous setting: A case study of the development of Yuendumu Old People's Programme. *Rural and Remote Health*. https://search.informit.org/doi/abs/10.3316/INFORMIT.396807774541079

Smulders, F. O., Vonk, J. A., Engel, M. S., & Christianen, M. J. (2017). Expansion and fragment settlement of the non-native seagrass *Halophila stipulacea* in a Caribbean bay. *Marine Biology Research*, *13*(9), 967–974. https://doi.org/10.1080/17451000.2017.1333620

Snow, R., Brien, M., Cherkiss, M. S., Wilkins, L., & Mazzotti, F. (2007). Dietary habits of the Burmese python, *Python molurus bivittatus*, in Everglades National Park, Florida. *The Herpetological Bulletin*. https://www.semanticscholar.org/paper/Dietary-habits-of-the-Burmese-python%2C-Python-in-Snow-Brien/86abadc2abaa11d9af9f75a866e147cded3f4de9

Sousa, A. M. M., Alves, V. D., Morais, S., Delerue-Matos, C., & Gonçalves, M. P. (2010). Agar extraction from integrated multitrophic aquacultured *Gracilaria vermiculophylla*: Evaluation of a microwave-assisted process using response surface methodology. *Bioresource Technology*, *101*(9), 3258–3267. https://doi.org/10.1016/j.biortech.2009.12.061

Sousa, R., Nogueira, J. G., Ferreira, A., Carvalho, F., Lopes-Lima, M., Varandas, S., & Teixeira, A. (2019). A tale of shells and claws: The signal crayfish as a threat to the pearl mussel *Margaritifera margaritifera* in Europe. *Science of The Total Environment*, *665*, 329–337. https://doi.org/10.1016/j.scitotenv.2019.02.094

Souty-Grosset, C., Anastácio, P. M., Aquiloni, L., Banha, F., Choquer, J., Chucholl, C., & Tricarico, E. (2016). The red swamp crayfish *Procambarus clarkii* in Europe: Impacts on aquatic ecosystems and human well-being. *Limnologica*, *58*, 78–93. https://doi.org/10.1016/j.limno.2016.03.003

Souza, A. O., Chaves, M. do P. S. R., Barbosa, R. I., & Clement, C. R. (2018). Local ecological knowledge concerning the invasion of Amerindian lands in the northern Brazilian Amazon by *Acacia mangium* (Willd.). *Journal of Ethnobiology and Ethnomedicine*, *14*(1), 33. https://doi.org/10.1186/s13002-018-0231-x

Spatz, D. R., Newton, K. M., Heinz, R., Tershy, B., Holmes, N. D., Butchart, S. H. M., & Croll, D. A. (2014). The Biogeography of Globally Threatened Seabirds and Island Conservation Opportunities: Seabird Conservation Opportunities. *Conservation Biology*, *28*(5), 1282–1290. https://doi.org/10.1111/cobi.12279

Spencer, C. N., McClelland, B. R., & Stanford, J. A. (1991). Shrimp stocking, salmon collapse, and eagle displacement. *BioScience*, *41*, 14–21. https://doi.org/doi.org/10.2307/1311536

Spencer, R.-J., Janzen, F. J., & Thompson, M. B. (2006). Counterintuitive density-dependent growth in a long-lived vertebrate after removal of nest predators. *Ecology*, *87*(12), 3109–3118. https://doi.org/10.1890/0012-9658(2006)87[3109:CDGIAL]2.0.CO;2

Sprengel, G., & Luchtenberg, H. (1991). Infection by endoparasites reduces maximum swimming speed of European smelt *Osmerus eperlanus* and European eel *Anguilla anguilla*. *Diseases of Aquatic Organisms*, *11*, 31–35. https://doi.org/10.3354/dao011031

Stafford, C. T. (1996). Hypersensitivity to Fire Ant Venom. *Annals of Allergy, Asthma & Immunology*, *77*(2), 87–99. https://doi.org/10.1016/S1081-1206(10)63493-X

Staples, G. W., & Cowie, R. H. (2001). *Hawai'i's invasive species*. Mutual Pub. https://agris.fao.org/agris-search/search.do?recordID=US201300094019

Staples, G. W., Herbst, D. R., & Imada, C. T. (2000). Survey of invasive or potentially invasive cultivated plants in Hawaii. *Bishop Museum Occasional Papers.* https://www.cabdirect.org/cabdirect/abstract/20177200474

Steadman, D. W. (1995). Prehistoric Extinctions of Pacific Island Birds: Biodiversity Meets Zooarchaeology. *Science*, *267*(5201), 1123–1131. https://doi.org/10.1126/science.267.5201.1123

Stefanowicz, A. M., Zubek, S., Stanek, M., Grześ, I. M., Rożej-Pabijan, E., Błaszkowski, J., & Woch, M. W. (2019). Invasion of *Rosa rugosa* induced changes in soil nutrients and microbial communities of coastal sand dunes. *Science of The Total Environment*, *677*, 340–349. https://doi.org/10.1016/j.scitotenv.2019.04.408

Steffani, C. N., & Branch, G. M. (2005). Mechanisms and consequences of competition between an alien mussel, *Mytilus galloprovincialis*, and an indigenous limpet, *Scutellastra argenvillei*. *Journal of Experimental Marine Biology and Ecology*, *317*(2), 127–142. https://doi.org/10.1016/j.jembe.2004.11.022

Steiner, S. C. C., & Willette, D. A. (2013). The invasive seagrass *Halophila stipulacea* (Hydrocharitaceae, Angiospermae) and its impact on the benthic landscape of Dominica, Lesser Antilles. *Los Angeles: Institute for Tropical Marine Ecology*. https://www.itme.org/reports/ITME_RReports32.pdf

Steiner, S. C. C., & Willette, D. A. (2015). The expansion of *Halophila stipulacea* (Hydrocharitaceae, Angiospermae) is changing the seagrass landscape in the commonwealth of Dominica, Lesser Antilles. *Caribb. Nat*, *22*, 1–19. https://www.semanticscholar.org/paper/The-Expansion-of-Halophila-stipulacea-Angiospermae)-Steiner-Willette/692f3fcfecb74171bc806ec6ebfaaa0903a6bb07

Stephens, K., Measey, J., Reynolds, C., & Le Roux, J. J. (2020). Occurrence and extent of hybridisation between the invasive Mallard Duck and native Yellow-billed Duck in South Africa. *Biological Invasions*, *22*(2), 693–707. https://doi.org/10.1007/s10530-019-02122-6

Strayer, D. L. (2009). Twenty years of zebra mussels: Lessons from the mollusk that made headlines. *Frontiers in Ecology and the Environment*, *7*(3), 135–141. https://doi.org/10.1890/080020

Strayer, D. L. (2010). Alien species in fresh waters: Ecological effects, interactions with other stressors, and prospects for the future. *Freshwater Biology*, *55*, 152–174. https://doi.org/10.1111/j.1365-2427.2009.02380.x

Suchanek, T. H., Geller, J. B., Kreiser, B. R., & Mitton, J. B. (1997). Zoogeographic Distributions of the Sibling Species *Mytilus galloprovincialis* and *M. trossulus* (Bivalvia: Mytilidae) and Their Hybrids in the North Pacific. *The Biological Bulletin*, *193*(2), 187–194. https://doi.org/10.2307/1542764

Sullivan, A., & York, A. M. (2021). Collective action for changing forests: A spatial, social-ecological approach to assessing participation in invasive plant management. *Global Environmental Change*, *71*, 102366. https://doi.org/10.1016/j.gloenvcha.2021.102366

Sullivan, A., York, A. M., An, L., Yabiku, S. T., & Hall, S. J. (2017). How does perception at multiple levels influence collective action in the commons? The case of *Mikania micrantha* in Chitwan, Nepal. *Forest Policy and Economics*, *80*, 1–10. https://doi.org/10.1016/j.forpol.2017.03.001

Sundaram, B., Krishnan, S., Hiremath, A. J., & Joseph, G. (2012). Ecology and Impacts of the Invasive Species, *Lantana camara*, in a Social-Ecological System in South India: Perspectives from Local Knowledge. *Human Ecology*, *40*(6), 931–942. https://doi.org/10.1007/s10745-012-9532-1

Swahn, J.-Ö. (2004). The cultural history of crayfish. *Bulletin Français de La Pêche et de La Pisciculture*, *372–73*, 243–251. https://www.semanticscholar.org/paper/The-cultural-history-of-crayfish-Swahn/50d7460af34e80408252200d4688d95b182479dd

Szabo, J. K., Khwaja, N., Garnett, S. T., & Butchart, S. H. (2012). Global patterns and drivers of avian extinctions at the species and subspecies level. *PLoS ONE*, *7*((10)), e47080. https://doi.org/10.1371/journal.pone.0047080

Talma, J., Kotze, J. D., Markovina, M., & Snijman, P. (2014). Joint Operations in Lake Victoria to reduce IUU fishing. *Illegal, Unreported and Unregulated (IUU) Fishing*, *7*((10)), e47080. https://www.fao.org/iuu-fishing/resources/detail/en/c/1132016/

Tambo, J. A., Kansiime, M. K., Rwomushana, I., Mugambi, I., Nunda, W., Mloza Banda, C., Nyamutukwa, S., Makale, F., & Day, R. (2021). Impact of fall armyworm invasion on household income and food security in Zimbabwe. *Food and Energy Security*, *10*(2), 299–312. https://doi.org/10.1002/fes3.281

Tambo, J. A., Uzayisenga, B., Mugambi, I., Bundi, M., & Silvestri, S. (2020). Plant clinics, farm performance and poverty alleviation: Panel data evidence from Rwanda. *World Development*, *129*, 104881. https://doi.org/10.1016/j.worlddev.2020.104881

Tano, S. A., Halling, C., Lind, E., Buriyo, A., & Wikström, S. A. (2015). Extensive spread of farmed seaweeds causes a shift from native to non-native haplotypes in natural seaweed beds. *Marine Biology*, *162*(10), 1983–1992. https://doi.org/10.1007/s00227-015-2724-7

Taylor, K. T., Maxwell, B. D., McWethy, D. B., Pauchard, A., Nuñez, M. A., & Whitlock, C. (2017). *Pinus contorta* invasions increase wildfire fuel loads and may create a positive feedback with fire. *Ecology*, *98*(3), 678–687. https://doi.org/10.1002/ecy.1673

Taylor, R. H. (1979). How the Macquarie Island parakeet became exinct. *New Zealand Journal of Ecology*. https://www.semanticscholar.org/paper/HOW-THE-MACQUARIE-ISLAND-PARAKEET-BECAME-EXTINCT-Taylor/0649ff286a87a60f54a137d6ff693183c311e708

Thieltges, D. W. (2005a). Benefit from an invader: American slipper limpet *Crepidula fornicata* reduces star fish predation on basibiont European mussels. *Hydrobiologia*, *541*(1), 241–244. https://doi.org/10.1007/s10750-004-4671-z

Thieltges, D. W. (2005b). Impact of an invader: Epizootic American slipper limpet *Crepidula fornicata* reduces survival and growth in European mussels. *Marine Ecology Progress Series*, *286*, 13–19. https://doi.org/10.3354/meps286013

Thieltges, D. W., & Buschbaum, C. (2007). Mechanism of an epibiont burden: *Crepidula fornicata* increases byssus thread production by *Mytilus edulis*. *Journal of Molluscan Studies*, *73*(1), 75–77. https://doi.org/10.1093/mollus/eyl033

Thieltges, D. W., Reise, K., Prinz, K., & Jensen, K. T. (2009). Invaders interfere with native parasite–host interactions. *Biological Invasions*, *11*(6), 1421–1429. https://doi.org/10.1007/s10530-008-9350-y

Thiengo, S. C., Maldonado, A., Mota, E. M., Torres, E. J. L., Caldeira, R., Carvalho, O. S., Oliveira, A. P. M., Simões, R. O., Fernandez, M. A., & Lanfredi, R. M. (2010). The giant African snail *Achatina fulica* as natural intermediate host of *Angiostrongylus cantonensis* in Pernambuco, northeast Brazil. *Acta Tropica*, *115*(3), 194–199. https://doi.org/10.1016/j.actatropica.2010.01.005

Thomsen, M. S. (2010). Experimental evidence for positive effects of invasive seaweed on native invertebrates *via* habitat-formation in a seagrass bed. *Aquatic Invasions*, *5*(4), 341–346. https://doi.org/10.3391/ai.2010.5.4.02

Timsina, B., Shrestha, B. B., Rokaya, M. B., & Münzbergová, Z. (2011). Impact of *Parthenium hysterophorus* L. invasion on plant species composition and soil properties of grassland communities in Nepal. *Flora – Morphology, Distribution, Functional Ecology of Plants*, *206*(3), 233–240. https://doi.org/10.1016/j.flora.2010.09.004

Tireman, H. (1916). Lantana in the forests of Coorg. *Indian Forester*, *42*(8), 384–392. https://www.i-scholar.in/index.php/indianforester/article/view/18030/0

Tirira, D., & Weksler, M. (2017). IUCN Red List of Threatened Species: *Nesoryzomys indefessus*. *IUCN Red List of Threatened Species*. https://www.iucnredlist.org/en

Tirira, D., & Weksler, M. (2019). IUCN Red List of Threatened Species: *Nesoryzomys darwini*. *IUCN Red List of Threatened Species*. https://www.iucnredlist.org/en

Tjaden, N. B., Caminade, C., Beierkuhnlein, C., & Thomas, S. M. (2018). Mosquito-borne diseases: Advances in modelling climate-change impacts. *Trends in Parasitology*, *34*(3), 227–245. https://doi.org/10.1016/j.pt.2017.11.006

Todesco, M., Pascual, M. A., Owens, G. L., Ostevik, K. L., Moyers, B. T., Hübner, S., Heredia, S. M., Hahn, M. A., Caseys, C., Bock, D. G., & Rieseberg, L. H. (2016). Hybridization and extinction. *Evolutionary Applications*, *9*(7), 892–908. https://doi.org/10.1111/eva.12367

Toscano, B. J., Newsome, B., & Griffen, B. D. (2014). Parasite modification of predator functional response. *Oecologia*, *175*, 345–352. https://doi.org/10.1007/s00442-014-2905-y

Tsai, H. C., Liu, Y. C., Kunin, C. M., Lee, S. S., Chen, Y. S., Lin, H. H., Tsai, T. H., Lin, W. R., Huang, C. K., Yen, M. Y., & Yen, C. M. (2001). Eosinophilic meningitis caused by *Angiostrongylus cantonensis*: Report of 17 cases. *The American Journal of Medicine*, *111*(2), 109–114. https://doi.org/10.1016/s0002-9343(01)00766-5

Tsopelas, P., Santini, A., Wingfield, M. J., & Wilhelm de Beer, Z. (2017). Canker Stain: A Lethal Disease Destroying Iconic Plane Trees. *Plant Disease*, *101*(5), 645–658. https://doi.org/10.1094/PDIS-09-16-1235-FE

Tuft, K., Legge, S., Frank, A. S. K., James, A. I., May, T., Page, E., Radford, I. J., Woinarski, J. C. Z., Fisher, A., Lawes, M. J., Gordon, I. J., & Johnson, C. N. (2021). Cats are a key threatening factor to the survival of local populations of native small mammals in Australia's tropical savannas: Evidence from translocation trials with *Rattus tunneyi*. *Wildlife Research*, *48*(7), 654–662. https://doi.org/10.1071/WR20193

Turbelin, A., & Catford, J. A. (2021). Invasive plants and climate change. In *Climate Change* (pp. 515–539). Elsevier. https://doi.org/10.1016/B978-0-12-821575-3.00025-6

Turot, O., Nyamsi, U., & Baitani, M. (2019). *The United Republic of Tanzania Socio-economic impact of fall armyworm on agricultural households in Iringa, Manyara and Morogoro Regions*. Food and Agriculture Organisation of the United Nations.

Tuwei, P., Odera, E. C., Kiprop, J., & Wanjiku, J. (2019). Management of *Prosopis juliflora* Invasion in Baringo County, Kenya through Utilization. *Journal of Economics and Sustainable Development*. https://doi.org/10.7176/JESD/10-10-09

Tyagi, V., Yadav, R., Sukumaran, D., & Veer, V. (2015). *Larvicidal activity of invasive weed Prosopis juliflora against mosquito species Anopheles* subpictus, *Culex quinquefasciatus and Aedes* aegypti. 4.

Uchii, K., Matsui, K., Iida, T., & Kawabata, Z. (2009). Distribution of the introduced cyprinid herpesvirus 3 in a wild population of common carp, *Cyprinus carpio* L. *Journal of Fish Diseases*, *32*(10), 857–864. https://doi.org/10.1111/j.1365-2761.2009.01064.x

Ueki, K., Ito, M., & Oki, Y. (1976). Waterhyacinth and its habitats in Japan. *Proceedings of the Fifth Asian-Pacific Weed Science Society Conference, Tokyo*, 424–428.

Upadhyay, B., Burra, D. D., Nguyen, T. T., & Wyckhuys, K. A. G. (2020). Caught off guard: Folk knowledge proves deficient when addressing invasive pests in Asian cassava systems. *Environment, Development and Sustainability*, *22*(1), 425–445. https://doi.org/10.1007/s10668-018-0208-x

U.S. Congress, Office of Technology Assessment. (1993). *Harmful Non-Indigenous Species in the United States*. Government Printing Office. https://www.princeton.edu/~ota/disk1/1993/9325/9325.PDF

Usher, M. B. (1988). Biological invasions of nature reserves: A search for generalisations. *Biological Conservation*, *44*(1–2), 119–135. https://doi.org/10.1016/0006-3207(88)90007-9

Vaarzon-Morel, P. (2010). Changes in Aboriginal perceptions of feral camels and of their impacts and management. *The Rangeland Journal*, *32*(1), 73. https://doi.org/10.1071/RJ09055

Valentine, P. C., Carman, M. R., Blackwood, D. S., & Heffron, E. J. (2007). Ecological observations on the colonial ascidian *Didemnum* sp. In a New England tide pool habitat. In *Journal of Experimental Marine Biology and Ecology* (Vol. 342, Issue 1, p. 109121). https://doi.org/10.1016/j.jembe.2006.10.021

Valentine, P. C., Collie, J. S., Reid, R. N., Asch, R. G., Guida, V. G., & Blackwood, D. S. (2007). The occurrence of the colonial ascidian *Didemnum* sp. On Georges Bank gravel habitat: Ecological observations and potential effects on groundfish and scallop fisheries. *Journal of Experimental Marine Biology and Ecology*, *342*, 179–181. https://doi.org/10.1016/j.jembe.2006.10.038

Valenzuela, A. E., Raya Rey, A., Fasola, L., Sáenz Samaniego, R. A., & Schiavini, A. (2013). Trophic ecology of a top predator colonizing the southern extreme of South America: Feeding habits of invasive American mink (*Neovison vison*) in Tierra del Fuego. *Mammalian Biology*, *78*(2), 104–110. https://doi.org/10.1016/j.mambio.2012.11.007

Vallet, C., Dauvin, J.-C., Hamon, D., & Dupuy, C. (2001). Effect of the Introduced Common Slipper Shell on the Suprabenthic Biodiversity of the Subtidal Communities in the Bay of Saint-Brieuc. *Conservation Biology*, *15*(6), 1686–1690. https://doi.org/10.1046/j.1523-1739.2001.99295.x

Van der Colff, D., Kumschick, S., Foden, W., & Wilson, J. R. U. (2020). Comparing the IUCN's EICAT and Red List to improve assessments of the impact of biological invasions. *NeoBiota*, *62*, 509–523. https://doi.org/10.3897/neobiota.62.52623

van Loon, M. P., Adjei-Nsiah, S., Descheemaeker, K., Akotsen-Mensah, C., van Dijk, M., Morley, T., van Ittersum, M. K., & Reidsma, P. (2019). Can yield variability be explained? Integrated assessment of maize yield gaps across smallholders in Ghana. *Field Crops Research*, *236*, 132–144. https://doi.org/10.1016/j.fcr.2019.03.022

van Riper III, C., van Riper, S. G., Goff, M. L., & Laird, M. (1986). The Epizootiology and Ecological Significance of Malaria in Hawaiian Land Birds. *Ecological Monographs*, *56*(4), 327–344. https://doi.org/10.2307/1942550

van Riper III, C., van Riper, S. G., & Hansen, W. R. (2002). Epizootiology and effect of avian pox on Hawaiian forest birds. *The Auk*, *119*(4), 929–942. https://doi.org/10.1093/auk/119.4.929

Vanderhoeven, S., Adriaens, T., Desmet, P., Strubbe, D., Backeljau, T., Barbier, Y., Brosens, D., Cigar, J., Coupremanne, M., De Troch, R., Eggermont, H., Heughebaert, A., Hostens, K., Huybrechts, P., Jacquemart, A.-L., Lens, L., Monty, A., Paquet, J.-Y., Prévot, C., … Groom, Q. (2017). Tracking Invasive Alien Species (TrIAS): Building a data-driven framework to inform policy. *Research Ideas and Outcomes*, *3*, e13414. https://doi.org/10.3897/rio.3.e13414

Vanderploeg, H. A., Nalepa, T. F., Jude, D. J., Mills, E. L., Holeck, K. T., Liebig, J. R., Grigorovich, I. A., & Ojaveer, H. (2002). Dispersal and emerging ecological impacts of Ponto-Caspian species in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*, *59*(7), 1209–1228. https://doi.org/10.1139/f02-087

Vaughn, C. C. (2018). Ecosystem services provided by freshwater mussels. *Hydrobiologia*, *810*(1), 15–27. https://doi.org/10.1007/s10750-017-3139-x

Vaz, A. S., Castro-Díez, P., Godoy, O., Alonso, Á., Vilà, M., Saldaña, A., Marchante, H., Bayón, Á., Silva, J. S., Vicente, J. R., & Honrado, J. P. (2018). An indicator-based approach to analyse the effects of non-native tree species on multiple cultural ecosystem services. *Ecological Indicators*, *85*, 48–56. https://doi.org/10.1016/j.ecolind.2017.10.009

Vaz, A. S., Kueffer, C., Kull, C. A., Richardson, D. M., Vicente, J. R., Kühn, I., Schröter, M., Hauck, J., Bonn, A., & Honrado, J. P. (2017). Integrating ecosystem services and disservices: Insights from plant invasions. *Ecosystem Services*, *23*, 94–107. https://doi.org/10.1016/j.ecoser.2016.11.017

Vázquez-Luis, M., Sanchez-Jerez, P., & Bayle-Sempere, J. T. (2008). Changes in amphipod (Crustacea) assemblages associated with shallow-water algal habitats invaded by *Caulerpa racemosa* var. *Cylindracea* in the western Mediterranean Sea. *Marine Environmental Research*, *65*(5), 416–426. https://www.academia.edu/21374333/Changes_in_amphipod_Crustacea_assemblages_associated_with_shallow_water_algal_habitats_invaded_by_Caulerpa_racemosa_var_cylindracea_in_the_western_Mediterranean_Sea

Venette, R. C. (2015). The challenge of modelling and mapping the future distribution and impact of invasive alien species. In *Pest risk modelling and mapping for invasive alien species* (USDA, pp. 1–17). CABI. https://doi.org/10.1079/9781780643946.0001

Verbrugge, L. N. H., Van Den Born, R. J. G., & Lenders, H. J. R. (2013). Exploring public perception of non-native species from a visions of nature perspective. *Environmental Management*, *52*(6), 1562–1573. https://doi.org/10.1007/s00267-013-0170-1

Verlaque, M., & Fritayre, P. (1994). Modifications des communautés algales méditerranéennes en présence de l'algue envahissante *Caulerpa taxifolio* (Vahl) C.Agardh. *Oceanologica Acta*, *17*(6), 659–672. https://archimer.ifremer.fr/doc/00099/21018/

Verstijnen, Y. (2019). Trophic relationships in Dutch reservoirs recently invaded by Ponto-Caspian species: Insights from fish trends and stable isotope analysis. *Aquatic Invasions*, *14*(2), 280–298. https://doi.org/10.3391/ai.2019.14.2.08

Vigliano, P. H., & Alonso, M. F. (2007). Salmonid introductions in Patagonia: A mixed blessing. In *Ecological and genetic implications of aquaculture activities* (Vol. 6, pp. 315–331). Springer. https://link.springer.com/chapter/10.1007/978-1-4020-6148-6_17

Vilà, M., Basnou, C., Pyšek, P., Josefsson, M., Genovesi, P., Gollasch, S., Nentwig, W., Olenin, S., Roques, A., Roy, D., & Hulme, P. E. (2010). How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Frontiers in Ecology and the Environment*, *8*(3), 135–144. https://doi.org/10.1890/080083

Vilà, M., Dunn, A. M., Essl, F., GÓmez-DÍaz, E., Hulme, P. E., Jeschke, J. M., NÚÑez, M. A., Ostfeld, R. S., Pauchard, A., Ricciardi, A., & Gallardo, B. (2021). Viewing Emerging Human Infectious Epidemics through the Lens of Invasion Biology. *BioScience*, *71*(7), 722–740. https://doi.org/10.1093/biosci/biab047

Vilà, M., & Hulme, P. (Eds.). (2017). *Impact of Biological Invasions on Ecosystem Services*. Springer International Publishing. https://www.springer.com/gb/book/9783319451190

Vilà, M., Williamson, M., & Lonsdale, M. (2004). Competition Experiments on Alien Weeds with Crops: Lessons for Measuring Plant Invasion Impact? *Biological Invasions*, *6*(1), 59–69. https://doi.org/10.1023/B:BINV.0000010122.77024.8a

Vilizzi, L., Tarkan, A. S., & Copp, G. H. (2015). Experimental Evidence from Causal Criteria Analysis for the Effects of Common Carp *Cyprinus carpio* on Freshwater Ecosystems: A Global Perspective. *Reviews in Fisheries Science & Aquaculture*, *23*(3), 253–290. https://doi.org/10.1080/23308249.2015.1051214

Villamagna, A. M., & Murphy, B. R. (2010). Ecological and socio-economic impacts of invasive water hyacinth (*Eichhornia crassipes*): A review. *Freshwater Biology*, *55*(2), 282–298. https://doi.org/10.1111/j.1365-2427.2009.02294.x

Vimercati, G., Kumschick, S., Probert, A. F., Volery, L., & Bacher, S. (2020). The importance of assessing positive and beneficial impacts of alien species. *NeoBiota*, *62*, 525–545. https://doi.org/10.3897/neobiota.62.52793

Vimercati, G., Probert, A. F., Volery, L., Bernardo-Madrid, R., Bertolino, S., Céspedes, V., Essl, F., Evans, T., Gallardo, B., Gallien, L., González-Moreno, P., Grange, M. C., Hui, C., Jeschke, J. M., Katsanevakis, S., Kühn, I., Kumschick, S., Pergl, J., Pyšek, P., … Bacher, S. (2022). The EICAT+ framework enables classification of positive impacts of alien taxa on native biodiversity. *PLoS Biology*, *20*(8), e3001729. https://doi.org/10.1371/journal.pbio.3001729

Vítková, M., Müllerová, J., Sádlo, J., Pergl, J., & Pyšek, P. (2017). Black locust (*Robinia pseudoacacia*) beloved and despised: A story of an invasive tree in Central Europe. *Forest Ecology and Management*, *384*, 287–302. https://doi.org/10.1016/j.foreco.2016.10.057

Vitousek, P. M., & Walker, L. R. (1989). Biological Invasion by *Myrica faya* in Hawai'i: Plant Demography, Nitrogen Fixation, Ecosystem Effects. *Ecological Monographs*, *59*(3), 247–265. https://doi.org/10.2307/1942601

Vitule, J. R. S., Freire, C. A., Vazquez, D. P., Nuñez, M. A., & Simberloff, D. (2012). Revisiting the potential conservation value of non-native species. *Conservation Biology: The Journal of the Society for Conservation Biology*, *26*(6), 1153–1155. https://doi.org/10.1111/j.1523-1739.2012.01950.x

Vogel, M., Remmert, H., & Smith, R. I. L. (1984). Introduced reindeer and their effects on the vegetation and the epigeic invertebrate fauna of South Georgia (subantarctic). *Oecologia*, *62*(1), 102–109. https://doi.org/10.1007/bf00377382

Volery, L., Blackburn, T. M., Bertolino, S., Evans, T., Genovesi, P., Kumschick, S., Roy, H. E., Smith, K. G., & Bacher, S. (2020). Improving the Environmental Impact Classification for Alien Taxa (EICAT): A summary of revisions to the framework and guidelines. *NeoBiota*, *62*, 547–567. https://doi.org/10.3897/neobiota.62.52723

Volery, L., Jatavallabhula, D., Scillitani, L., Bertolino, S., & Bacher, S. (2021). Ranking alien species based on their risks of causing environmental impacts: A global assessment of alien ungulates. *Global Change Biology*, *27*(5), 1003–1016. https://doi.org/10.1111/gcb.15467

Vorsino, A. E., Fortini, L. B., Amidon, F. A., Miller, S. E., Jacobi, J. D., Price, J. P., Iii, S. 'Ohukani'ohi'a G., & Koob, G. A. (2014). Modeling Hawaiian Ecosystem Degradation due to Invasive Plants under Current and Future Climates. *PLoS ONE*, *9*(5), e95427. https://doi.org/10.1371/journal.pone.0095427

Vyn, R. J. (2022). *Estimated Annual Expenditures on Invasive Species by Canadian Municipalities: 2021 National Survey Results* (p. 35). Invasive Species Centre. https://www.invasivespeciescentre.ca/wp-content/uploads/2022/04/Final-Report-2021-National-Survey-Results-Final-Version.pdf

Wagner, W. L., Herbst, D. R., & Sohmer, S. H. (1990). *Manual of the flowering plants of Hawaii*. University of Hawaii Press : Bishop Museum Press.

Walsh, F., Christophersen, P., & Mcgregor, S. (2013). Indigenous perspectives on biodiversity. In S. Morton, A. Sheppart, & W. M. Lonsdale (Eds.), *Biodiversity* (pp. 81–100). CSIRO Publishing. http://www.publish.csiro.au/ebook/download/pdf/6967#page=88

Walsh, S. K., Pender, R. J., Junker, R. R., Daehler, C. C., Morden, C. W., & Lorence, D. H. (2019). Pollination biology reveals challenges to restoring populations of *Brighamia insignis* (Campanulaceae), a critically endangered plant species from Hawai'i. *Flora*, *259*, 151448. https://doi.org/10.1016/j.flora.2019.151448

Walter, K. J., & Armstrong, K. V. (2014). Benefits, threats and potential of *Prosopis* in South India. *Forests, Trees and Livelihoods*, *23*(4), 232–247. https://doi.org/10.1080/14728028.2014.919880

Walton, W. C., MacKinnon, C., Rodriguez, L. F., Proctor, C., & Ruiz, G. M. (2002). Effect of an invasive crab upon a marine fishery: Green crab, *Carcinus maenas*, predation upon a venerid clam, *Katelysia scalarina*, in Tasmania (Australia). *Journal of Experimental Marine Biology and Ecology*, *272*(2), 171–189. https://doi.org/10.1016/S0022-0981(02)00127-2

Warner, R. E. (1968). The Role of Introduced Diseases in the Extinction of the Endemic Hawaiian Avifauna. *The Condor*, *70*(2), 101–120. https://doi.org/10.2307/1365954

Waser, A. M., Deuzeman, S., Kangeri, A. K. wa, van Winden, E., Postma, J., de Boer, P., van der Meer, J., & Ens, B. J. (2016). Impact on bird fauna of a non-native oyster expanding into blue mussel beds in the Dutch Wadden Sea. *Biological Conservation*, *202*, 39–49. https://doi.org/10.1016/j.biocon.2016.08.007

Watari, Y., Komine, H., Angulo, E., Diagne, C., Ballesteros-Mejia, L., & Courchamp, F. (2021). First synthesis of the economic costs of biological invasions in Japan. *NeoBiota*, *67*, 79–101. https://doi.org/10.3897/neobiota.67.59186

Wells, K., O'Hara, R. B., Morand, S., Lessard, J.-P., & Ribas, A. (2015). The Importance of Parasite Geography and Spillover Effects for Global Patterns of Host-Parasite Associations in Two Invasive Species. *Diversity and Distributions*, *21*(4), 477–486. https://doi.org/10.1111/ddi.12297

Wheeler, R., & Priddel, D. (2009). The impact of introduced predators on two threatened prey species: A case study from western New South Wales. *Ecological Management & Restoration*, *10*(1), S117–S123. https://doi.org/10.1111/j.1442-8903.2009.00457.x

Whinam, J., Fitzgerald, N., Visoiu, M., & Copson, G. (2014). Thirty years of vegetation dynamics in response to a fluctuating rabbit population on sub-Antarctic Macquarie Island. *Ecological Management & Restoration*, *15*(1), 41–51. https://doi.org/10.1111/emr.12076

White, S. M., Bullock, J. M., Hooftman, D. A. P., & Chapman, D. S. (2017). Modelling the spread and control of *Xylella fastidiosa* in the early stages of invasion in Apulia, Italy. *Biological Invasions*, *19*(6), 1825–1837. https://doi.org/10.1007/s10530-017-1393-5

Whitlow, W. L. (2010). Changes in survivorship, behavior, and morphology in native soft-shell clams induced by invasive green crab predators. *Marine Ecology*, *31*(3), 418–430. https://doi.org/10.1111/j.1439-0485.2009.00350.x

Whitworth, D. L., Carter, H. R., & Gress, F. (2013). Recovery of a threatened seabird after eradication of an introduced predator: Eight years of progress for Scripps's murrelet at Anacapa Island, California. *Biological Conservation*, *162*, 52–59. https://doi.org/10.1016/j.biocon.2013.03.026

Wiles, G. J., Bart, J., Beck, R. E., & Aguon, C. F. (2003). Impacts of the Brown Tree Snake: Patterns of Decline and Species Persistence in Guam's Avifauna. *Conservation Biology*, *17*(5), 1350–1360. https://doi.org/10.1046/j.1523-1739.2003.01526.x

Williams, L. K., Shaw, J. D., Sindel, B. M., Wilson, S. C., & Kristiansen, P. (2018). Longevity, growth and community ecology of invasive *Poa annua* across environmental gradients in the subantarctic. *Basic and Applied Ecology*, *29*, 20–31. https://doi.org/10.1016/j.baae.2018.02.003

Williams, S. L. (1995). Malleefowl as a flagship for conservation on farms in the Murray Mallee of South Australia. *Nature Conservation*, *4*, 316–320.

Willson, J. D. (2017). Indirect effects of invasive Burmese pythons on ecosystems in southern Florida. *Journal of Applied Ecology*, *54*(4), Article 4. https://doi.org/10.1111/1365-2664.12844

Wilson, E. E., Sidhu, C. S., LeVAN, K. E., & Holway, D. A. (2010). Pollen foraging behaviour of solitary Hawaiian bees revealed through molecular pollen analysis. *Molecular Ecology*, *19*(21), 4823–4829. https://doi.org/10.1111/j.1365-294X.2010.04849.x

Wilson, G., Desai, A. A., Sim, D. A., & Linklater, W. L. (2013). The influence of the invasive weed *Lantana camara* on elephant habitat use in Mudumalai Tiger Reserve, southern India. *Journal of Tropical Ecology*, *29*(3), 199–207. https://doi.org/10.1017/S0266467413000205

Wilson, G., Gruber, M. A. M., & Lester, P. J. (2014). Foraging Relationships Between Elephants and *Lantana camara* Invasion in Mudumalai Tiger Reserve, India. *Biotropica*, *46*(2), 194–201. https://doi.org/10.1111/btp.12094

Wilson, S. J., & Ricciardi, A. (2009). Epiphytic macroinvertebrate communities on Eurasian watermilfoil (*Myriophyllum spicatum*) and native milfoils *Myriophyllum sibericum* and *Myriophyllum alterniflorum* in eastern North America. *Canadian Journal of Fisheries and Aquatic Sciences*, *66*(1), 18–30. https://doi.org/10.1139/F08-187

Wise, R. M., van Wilgen, B. W., Hill, M. P., Schulthess, F., Tweddle, D., Chabi-Olaye, A., & Zimmermann, H. G. (2007). *The Economic Impact and Appropriate Management of Selected Invasive Alien Species on the African Continent* (CSIR Report CSIR/NRE/RBSD/ER/2007/0044/C; p. 64). https://stg-wedocs.unep.org/bitstream/handle/20.500.11822/19807/The_Economic_Impact_and_Appropriate_Management.pdf?sequence=1

Wise, R. M., van Wilgen, B. W., & Le Maitre, D. C. (2012). Costs, benefits and management options for an invasive alien tree species: The case of mesquite in the Northern Cape, South Africa. *Journal of Arid Environments*, *84*, 80–90. https://doi.org/10.1016/j.jaridenv.2012.03.001

Witt, A. B. R., Shackleton, R. T., Beale, T., Nunda, W., & Van Wilgen, B. W. (2019). Distribution of invasive alien *Tithonia* (Asteraceae) species in eastern and southern Africa and the socio-ecological impacts of *T. diversifolia* in Zambia. *Bothalia*, *49*(1), a2356. https://doi.org/10.4102/abc.v49i1.2356

Witte, F., Goldschmidt, T., Wanink, J., van Oijen, M., Goudswaard, K., Witte-Maas, E., & Bouton, N. (1992). The destruction of an endemic species flock: Quantitative data on the decline of the haplochromine cichlids of Lake Victoria. *Environmental Biology of Fishes*, *34*(1), Article 1. https://doi.org/10.1007/BF00004782

Witte, F., Kishe-Machumu, M. A., Mkumbo, O. C., Wanink, J. H., Goudswaard, P. C., Van Rijssel, J. C., & van Oijen, M. J. (2013). The fish fauna of Lake Victoria during a century of human induced perturbations. *Proc. Fourth Int. Conf. On African Fish and Fisheries, Addis Ababa, Ethiopia*, 22–26. https://www.semanticscholar.org/paper/The-Fish-Fauna-of-Lake-Victoria-during-a-Centruy-of-Witte-Kish-Machumu/f0de79e7cb8a1f210b4f30b6aca76d8e9c658fe3

Woinarski, J. C. Z., Burbidge, A. A., & Harrison, P. L. (2015). Ongoing unraveling of a continental fauna: Decline and extinction of Australian mammals since European settlement. *Proceedings of the National Academy of Sciences*, *112*(15), 4531–4540. https://doi.org/10.1073/pnas.1417301112

Woinarski, J. C. Z., Legge, S., Fitzsimons, J. A., Traill, B. J., Burbidge, A. A., Fisher, A., Firth, R. S., Gordon, I. J., Griffiths, A. D., & Johnson, C. N. (2011). The disappearing mammal fauna of northern Australia: Context, cause, and response. *Conservation Letters*, *4*(3), 192–201. https://conbio.onlinelibrary.wiley.com/doi/epdf/10.1111/j.1755-263X.2011.00164.x

Wolff, W. J., & Reise, K. (2002). Oyster Imports as a Vector for the Introduction of Alien Species into Northern and Western European Coastal Waters. In E. Leppäkoski, S. Gollasch, & S. Olenin (Eds.), *Invasive Aquatic Species of Europe. Distribution, Impacts and Management* (pp. 193–205). Springer Netherlands. https://doi.org/10.1007/978-94-015-9956-6_21

Wood, J. R., Alcover, J. A., Blackburn, T. M., Bover, P., Duncan, R. P., Hume, J. P., Louys, J., Meijer, H. J. M., Rando, J. C., & Wilmshurst, J. M. (2017). Island extinctions: Processes, patterns, and potential for ecosystem restoration. *Environmental Conservation*, *44*(4), 348–358. https://doi.org/10.1017/S037689291700039X

World Health Organization & Convention on Biological Diversity. (2015). *Connecting global priorities: Biodiversity and human health: a state of knowledge review*. World Health Organization. https://apps.who.int/iris/handle/10665/174012

Wright, A., Yap, M., Jones, R., Richardson, A., Davis, V., & Lovett, R. (2021). Examining the Associations between Indigenous Rangers, Culture and Wellbeing in Australia, 2018–2020. *International Journal of Environmental Research and Public Health*, *18*(6), 3053. https://www.mdpi.com/1660-4601/18/6/3053

Wright, J. T., Gribben, P. E., Byers, J. E., & Monro, K. (2012). Invasive ecosystem engineer selects for different phenotypes of an associated native species. *Ecology*, *93*(6), 1262–1268. https://doi.org/10.1890/11-1740.1

Xu, Y., Huang, J., Zhou, A., & Zeng, L. (2012). Prevalence of *Solenopsis invicta* (Hymenoptera: Formicidae) Venom Allergic Reactions in Mainland China. *Florida Entomologist*, *95*(4), 961–965. https://doi.org/10.1653/024.095.0421

Yahya, B. M., Yahya, S. A., Mmochi, A. J., & Jiddawi, N. S. (2020). The trophic structure of fish in seaweed farms, and adjacent seagrass and coral habitats in Zanzibar, Tanzania. *Western Indian Ocean Journal of Marine Science*, *19*(2), Article 2. https://doi.org/10.4314/wiojms.v19i2.2

Yang, X., Wyckhuys, K. A. G., Jia, X., Nie, F., & Wu, K. (2021). Fall armyworm invasion heightens pesticide expenditure among Chinese smallholder farmers. *Journal of Environmental Management*, *282*, 111949. https://doi.org/10.1016/j.jenvman.2021.111949

Yorio, P. M., Suarez, N. M., Kasinsky Aguilera, L. T., Pollicelli, M., Ibarra, C., & Gatto, A. J. (2020). *The introduced green crab (Carcinus maenas) as a novel food resource for the opportunistic kelp gull (*Larus dominicanus*) in Argentine Patagonia*. http://dx.doi.org/10.3391/ai.2020.15.1.10

Yule, A. M., Barker, I. K., Austin, J. W., & Moccia, R. D. (2006). Toxicity of *Clostridium botulinum* type E neurotoxin to Great Lakes fish: Implications for avian botulism. *Journal of Wildlife Diseases*, *42*(3), 479–493. https://doi.org/10.7589/0090-3558-42.3.479

Zalba, S. M., Cuevas, Y. A., & Boó, R. M. (2008). Invasion of *Pinus halepensis* Mill. Following a wildfire in an Argentine grassland nature reserve. *Journal of Environmental Management*, *88*(3), 539–546. https://doi.org/10.1016/j.jenvman.2007.03.018

Zengeya, T. A., Ivey, P., Woodford, D. J., Weyl, O., Novoa, A., Shackleton, R., Richardson, D., & Van Wilgen, B. (2017). Managing conflict-generating invasive species in South Africa: Challenges and trade-offs. *Bothalia*, *47*(2), a2160. https://doi.org/10.4102/abc.v47i2.2160

Zenni, R. D., Essl, F., García-Berthou, E., & McDermott, S. M. (2021). The economic costs of biological invasions around the world. *NeoBiota*, *67*, 1–9. https://doi.org/10.3897/neobiota.67.69971

Zenni, R. D., & Nuñez, M. A. (2013). The elephant in the room: The role of failed invasions in understanding invasion biology. *Oikos*, *122*(6), 801–815. https://doi.org/10.1111/j.1600-0706.2012.00254.x

Zhang, R., Li, Y., Liu, N., & Porter, S. D. (2007). An Overview of the Red Imported Fire Ant (Hymenoptera: Formicidae) in Mainland China. *Florida Entomologist*, *90*(4), 723–731. https://doi.org/10.1653/0015-4040(2007)90[723:AOOTRI]2.0.CO;2

Ziller, S. R., Reaser, J. K., Neville, L. E., & Brandt, K. (Eds.). (2005). *Invasive alien species in South America: National reports of resources*. Global Invasive Species Programme.

Znój, A., Chwedorzewska, K. J., Androsiuk, P., Cuba-Diaz, M., Giełwanowska, I., Koc, J., Korczak-Abshire, M., Grzesiak, J., & Zmarz, A. (2017). Rapid environmental changes in the Western Antarctic Peninsula region due to climate change and human activity. *Applied Ecology and Environmental Research*, *15*(4), 525–539. http://eprints.ibb.waw.pl/1364/1/Znoj%20et%20al..pdf

Chapter **5**

MANAGEMENT; CHALLENGES, OPPORTUNITIES AND LESSONS LEARNED[1]

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Chapter 5

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Chapter **5**

MANAGEMENT; CHALLENGES, OPPORTUNITIES AND LESSONS LEARNED

EXECUTIVE SUMMARY

1 **Substantial knowledge, options and experience in management of biological invasions exist at the regional, national and local levels *(well established)* {5.2, 5.3, 5.4, 5.5, 5.6}.** Key challenges for the effective implementation of biological invasion management include developing appropriate policies and regulations to support management {5.6.2.1}, building management capability and capacity {5.6.2.4} **(Table 5.11)**, fostering collaborative governance to assist stakeholder engagement within cultural contexts {5.2.1} and developing and implementing processes to manage cross-jurisdictional and transboundary issues **(Table 5.11)**. Addressing these challenges can ensure effective implementation of management strategies *(well established)* {5.6.2}.

2 **Options for management of biological invasions include pathway, species-based and site- or ecosystem-based approaches *(well established)* {5.1.1}. When implemented, these approaches will help mitigate impacts of invasive alien species and enhance ecosystem resilience *(well established)* {5.3, 5.5}.** Pathway management is a major component of prevention *(well established)* {5.5.1}. Species-based management, which includes surveillance, detection, eradication, containment and control, has been effective in many contexts *(well established)* {5.5.2, 5.5.3, 5.5.4, 5.5.5} **(Figure 5.1)**. Species-based and site-based approaches, such as species removal through adaptive management and ecosystem restoration, are likely to enhance cost-effective improvement of nature, nature's contributions to people and good quality of life *(well established)* {5.5.1, 5.3.2, 5.5.3, 5.5.6}. Integrated pathway, species-based and site-based management through engagement of stakeholders and Indigenous Peoples and local communities, can optimize management outcomes *(established but incomplete)* {5.2.1, 5.6.2.1}.

3 **Prevention, where possible, together with preparedness through surveillance and rapid intervention, is a cost-effective long-term biosecurity approach *(well established)* {5.5.1, 5.5.2}.** Preventive actions early in the biological invasion process decrease new species' arrivals and interception rates at the border and post-border *(well established)* {5.5.1}. Pre-border biosecurity planning and the implementation of prevention strategies through anthropogenic pathways is cost-effective in reducing biological invasions *(well established)* {5.5.2, 5.6.3.3}. Prevention is particularly effective for managing biological invasions in marine and connected water systems (e.g., ballast water and biofouling management) *(well established)* {5.5.1}. Safe trade that avoids biological invasions is supported by international sanitary, phytosanitary and animal health standards *(well established)* {5.3.1.1}. Preparedness based on surveillance, early detection and rapid response systems can quickly support cost-effective delimitation and eradication of newly established alien species when possible *(well established)* {5.5.1, 5.5.2, 5.5.3}. General surveillance systems for new alien species (e.g., through citizen science) are the most cost-effective approach to preparedness *(established but incomplete)* {5.4.2.2}. Many policies exist to support prevention of movement of invasive alien species pre-border, at the border and post-border *(well established)* {5.6.3.3}.

4 **Eradicating invasive alien species can be highly cost-effective on small islands or similar biogeographically isolated habitats of high biodiversity value and for localized and easily delimited invasive alien species *(established but incomplete)* {5.5.3}.** Eradication methods for invasive alien species on islands and similar isolated habitats are well developed, particularly for animals, and provide good examples of successful management *(well established)* {5.5.3}. Eradication other than on relatively small islands is typically only effective for highly localized incursions that spread slowly and/or where detection of invasive alien species' presence and delimitation is easy, where re-introductions are unlikely, and where effective removal techniques are available {5.5.3}. For invasive alien plant eradication, tested decision support tools exist *(well established)* {5.5.3}. Eradication programmes with community support on inhabited islands can become more challenging as island size increases *(established but incomplete)* {5.5.3}. Successful eradication of invasive alien species directly benefits good quality of life *(well established)* {5.5.3}.

5 **Effective management tools and technologies have been developed for prevention, preparedness and intervention *(well established)* {5.4}. New technologies are also being developed to improve**

complementary management approaches including ecosystem restoration *(well established)* {5.4}. Many platform-based tools exist or are under development including a) surveillance tools using remote sensing, sensory and genetic data capture and analytics, b) lab-based and in field diagnostics, c) robotic detection and intervention, d) biological control and e) adaptive management and ecosystem restoration *(well established)* {5.4.4} **(Table 5.6, Table 5.7)**. Smartphone-based data capture and analysis have game-changed affordability and adoption of digital invasive alien species management tools *(well established)* {5.4.4}. Developing novel technologies using a transparent precautionary approach in consultation with stakeholders, Indigenous Peoples and local communities, and regulators builds social licence and avoids unintended consequences *(well established)* {5.4.3.2}.

6 Many decision-support approaches, tools and methods exist to assist choice of management actions *(well established)* {5.2}. Decision-support approaches, tools and methods include scenarios and modelling, evidence-based tools that can identify hazards, prioritize pathways, species and sites for action *(well established)* {5.2.2.1}. Decision-support systems support transparency, adaptability and repeatability, through broad stakeholder community engagement, learning and endorsement of actions *(well established)* {5.2.2, 5.6.3.2}. Evidence- and consultation-based, quantitative and qualitative decision-support tools exist as standards and frameworks, and are supported by scenario and modelling platforms *(well established)* {5.2.2, 5.2.2.1, 5.6.3.2}.

7 Adaptive management, wherever possible led by stakeholders and Indigenous Peoples and local communities, promotes wide acceptance and capacity-building, and optimization of management success *(well established)* {5.2, 5.3, 5.6}. Failure to engage with Indigenous Peoples and local communities, especially those who are adapted to and use invasive alien species, in planning and implementing management actions can reduce good quality of life through loss of livelihoods, marginalization and/or gender inequity *(well established)* {5.2.1, 5.3.1.3, 5.4.4.2, 5.6.1.1, 5.6.1.2}. Broad and inclusive engagement improves planning, decision-making and undertaking management actions *(established but incomplete)* {5.2.1, 5.5.1.2}. This engagement is best achieved through partnerships around co-design, co-development and co-implementation and social learning *(established but incomplete)* {5.2.1, 5.4.4.3, 5.6.2.1}. Management programmes are most successful when their goal stretches beyond invasive alien species suppression to include restoring ecosystem resilience and nature's contributions to people *(established but incomplete)* {5.5.6}.

8 Though gaps exist in knowledge, data and management implementation, collective management actions can still proceed supported by stakeholders and Indigenous and local knowledge under a precautionary approach *(well established)* {5.2.2.1, 5.2.2.3, 5.2.2.4, 5.3.3, 5.4.4} (Box 5.13). Many sources of open-access data and analytical tools already exist to support capacity-building, priority setting, monitoring and management. However, there are many knowledge and data gaps which impede the development and implementation of pathway, species-based and site/ecosystem-based management approaches *(well established)* {5.6.2.1, 5.6.2.2, 5.6.2.3}. Despite this, effective decision-making and adaptive management programmes can still lead to successful outcomes if supported by stakeholders and Indigenous and local knowledge *(well established)* {5.2, 5.4, 5.6}. Addressing these knowledge and data gaps and uncertainty (e.g., on global change impacts) will improve management decisions and outcomes *(established but incomplete)* {5.2, 5.4, 5.6}. Improvements can be achieved by better capturing, sharing, integrating and analysing data in a manner that supports decision-making *(well established)* {5.2., 5.4}.

9 International and cross-sectoral collaboration through capacity-building networks and research and management partnerships improves transboundary management of biological invasions *(well established)* {5.6.3.1, 5.3.1}. The establishment of international networks between governments, scientists, non-governmental organizations, industries, relevant stakeholders, and Indigenous Peoples and local communities can help in the implementation of transboundary and cross-sectoral management of biological invasions *(well established)* {5.6.3.1}. International networks and partnerships help collective action, which may lead to societally acceptable and feasible management strategies and outcomes *(well established)* {5.6.3.1, 5.3.1}.

10 Failure to effectively manage biological invasions can result from data gaps, lack of awareness and societal, capacity, capability, resource and policy-related constraints especially in developing countries *(well established)* {5.3.1, 5.6.2.1, 5.6.2.2, 5.6.2.4}. Goals for the management of biological invasions are often not achieved even after considerable efforts. Gaps in data and knowledge on the distribution and spread of invasive alien species and lack of information on direct and indirect drivers of change facilitating biological invasions impede management in certain regions (e.g., parts of Asia, Africa and America) *(well established)* {5.6.2.1, 6.6.1.4}. Failures in management success can also be attributed to conflicting interests and values, lack of public awareness and understanding of impacts *(well established)* {5.6.2.1, 5.6.2.4}, inadequate policies and governance, poor capability and capacity, lack of resources, poor knowledge on modern tools and techniques and inefficiency to utilize them, and divergent public perspectives on individual species *(well established)* {5.3.1, 5.6.2.1, 5.6.2.2, 5.6.2.4}. Policy generally fails to address collective management for conflict species, for which there are positive and negative impacts of the invasive alien species on different stakeholders or sectors *(well established)* {5.6.1.2}.

11 Management of biological invasions that takes into account global change can also improve climate change resilience in ecosystems impacted by invasive alien species *(established but incomplete)* {5.6.1.3}. Effective management of invasive alien species can increase the long-term functional resilience of threatened ecosystems and habitats to climate change. Conversely, extreme climate events increase ecosystem susceptibility to invasive alien species. In such situations, rapid response through targeted adaptive management practices and ecosystem restoration supported by monitoring and collective decision-making can maintain benefits from existing management programmes *(established but incomplete)* {5.6.1.3}.

12 Long-term monitoring effectively supports management actions and sustains beneficial outcomes *(well established)* {5.4.4.2, 5.4.4.3, 5.5.3}. Long-term monitoring can be used to assess efficacy and outcomes of management actions and ensure sustained control of invasive alien species and ecosystem restoration *(well established)* {5.5.6, 5.5.7}. Long-term monitoring is also important for early detection of reinvasion *(well established)* {5.5.3}. These long-term efforts and strategies are best supported by cost-benefit, cost-effectiveness and risk analyses {5.2.2.1, 5.5.3} that consider benefits to Indigenous Peoples and local communities *(established but incomplete)* {5.4.4.2}.

5.1 INTRODUCTION

"Management" of biological invasions is conceptualized in at least two ways in different parts of the world. In the context of the invasive alien species assessment of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), this term encompasses any activity or action undertaken to directly or indirectly prevent or mitigate negative impacts of invasive alien species. This includes pathway, species-based and site- or ecosystem-based management activities (**Glossary**).

Management of biological invasions is a global concern, and thus was an explicit element of Aichi Target 9 of the Strategic Plan for Biodiversity 2011-2020, and is a main focus of Target 6 of the Kunming-Montreal Global Biodiversity Framework (CBD, 2022a; **Chapter 1**, **section 1.1**, **Box 1.2**) and Target 15.8 of the 2030 Agenda for Sustainable Development. In this context, this chapter provides policymakers and practitioners with a range of options and scenarios where management actions can be optimally applied **(Box 5.1)**.

Chapter 5 has been written based on a comprehensive literature and technical review of tools, strategies, challenges and key outcomes of management of biological invasions. It reflects the current state of management of biological invasions and its implications as covered in existing peer-reviewed and grey literature available to the assessment team.

Section 5.1 provides an overview of the invasion continuum, the management objectives and approaches applicable at

Box 5 1 **Rationale of the chapter.**

Chapter 5 assesses the efficacy of past and current programmes and tools for the local, national and global prevention (**Glossary**) and management of biological invasions and the impacts of invasive alien species. In particular, the chapter reviews past experience with: (a) preventing the spread of invasive alien species including the role of trade and economic development; (b) the precautionary approach (**Glossary**) in preventing and managing biological invasions and the efficacy of risk assessment as a tool for their management; (c) the adoption of biosecurity approaches (**Glossary**); (d) managing complexity and intersectoral conflicts, including on the use of an invasive alien species depending on contexts and values; (e) uses of social media and citizen science (**Glossary**) for the detection of invasive alien species and prevention and management of biological invasions; (f) eradicating or managing invasive alien species, including control options such as precision application of pesticides, baits, biological control and "gene drive" technology (**Glossary**); (g) capacities of different countries to manage biological invasions, and barriers to the uptake of tools; (h) managing biological invasions in protected areas, including wetlands and biosphere reserves; (i) managing biological communities invaded by alien species, considering co-existence, including direct and indirect interspecific interactions; and (j) managing biological invasions in the context of the complex interactions between alien species, their invasion status and climate change.

Guiding questions:

- What are the decision-support processes, tools and frameworks available for prevention and management of invasive alien species? (**section 5.2**)
- How best to target invasive alien species management through pathway, species-based and site-based or ecosystem-based management options under different scenarios such as management goals, status of invasion and the socio-economic context? (**section 5.3**)

Box 5 1

- How to use databases, modern tools, emerging technologies and scenarios and modelling more effectively in detecting, preventing and managing invasive alien species? (**section 5.4**)
- How effective are the various management options at various steps in the invasion process? (**section 5.5**)
- How can international networks assist in the prevention and management of invasive alien species? What role can regional partnerships play? (**section 5.6**)
- How critical is stakeholder participation including of Indigenous Peoples and local communities in management success? (**section 5.5**)
- What are the obstacles to the uptake of invasive alien species prevention and management implementation? (**section 5.6**)
- What methods are available for managing invasive alien species on islands and similar habitats (**Glossary**) of high biodiversity value? (**sections 5.5** and **5.6**)

Keywords:
Biological control, containment, eradication, invasion stages, monitoring, pathway, prevention, surveillance, site-based management, species-based management.

different phases of invasion and suited to different biomes. It discusses the challenges and opportunities of management.

Section 5.2 focuses on decision-making frameworks for identifying and prioritizing targets and options in pathway, species-based and site-based management. It reviews methodologies and tools available, how these can be used to prioritize targets and addresses uncertainty considerations in decision-making.

Section 5.3 assesses what pathway, species-based and site-based management strategies are and when to implement each, and integrates these for application at local to regional scales. Practical examples of these approaches are nested within a sociological and socio-economic context to enhance good quality of life, with a focus on protected areas and islands.

Section 5.4 presents a summary of management approaches, frameworks, platforms, scenarios and models, tools and technologies for current and potential application of management actions. It explores how new technologies are deployed and the efficacy of various tools and their future potential to improve management actions.

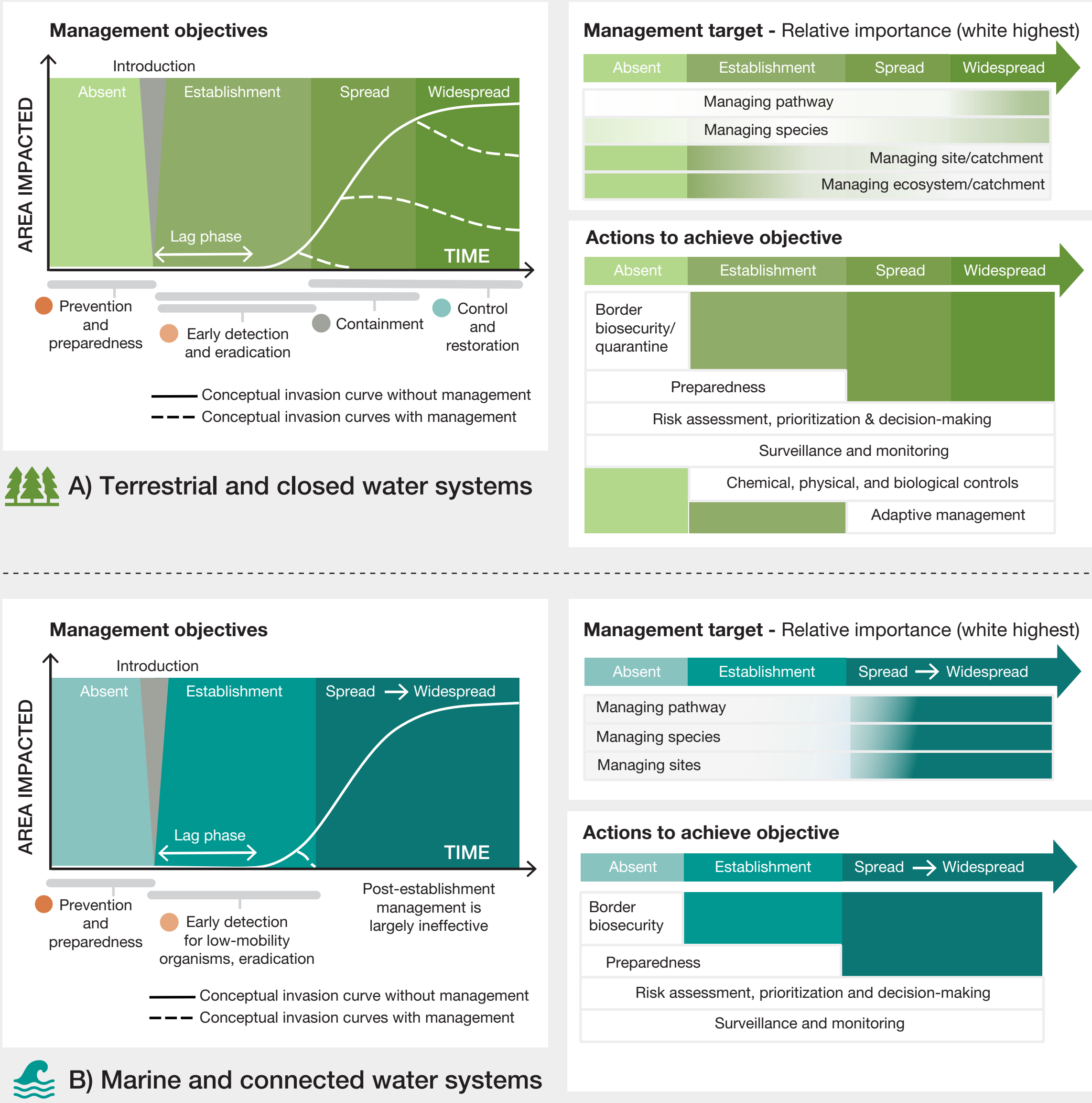
Section 5.5 assesses examples of successful and unsuccessful management approaches and examines how evidence-based decision-making and modern tools and technologies have brought successful management outcomes. It also provides evidence on management costs.

Section 5.6 summarizes the challenges in achieving effective management of biological invasions. It emphasizes context dependency and perspectives of various stakeholders and Indigenous Peoples and local communities on invasive alien species and related social conflict and discusses the knowledge gaps, lack of expertise and uncertainty which constrain effective management.

Section 5.7 provides a short conclusion.

5.1.1 Biological invasion management continuum

This chapter explores solutions to mitigate the impacts of invasive alien species (**Glossary**; **Chapter 1**, **Figure 1.1**) across biomes, species and regions. Any successful management action to prevent spread and ameliorate current or future potential impacts of an invasive alien species is built on a co-developed overarching management objective that goes beyond targeting one or more of the invasive alien species. The progression of a biological invasion by a species is generally divided into four stages, with a range of optimal management strategies which vary along this biological invasion continuum (**Figure 5.1**; **Chapter 1**, **Figure 1.8**). The management-invasion continuum (often called the "Invasion Curve", see **Glossary**) can be visually conceptualized to show the changes in management objectives and focus through each of the stages of the biological invasion process **(Figure 5.1)**. "Introduction" refers to the many introductions from intermittent or continuous propagule pressure (**Glossary**). Management approaches and responses may vary depending on whether the affected ecosystem is terrestrial or closed water systems, such as catchment basins, coastal systems and salt marshes, or an open water system (e.g., marine, brackish and water connected systems). Therefore, management-invasion continuum is presented here for these two scenarios; one for terrestrial and closed water ecosystems **(Figure 5.1A)** and another for an open water system **(Figure 5.1B)**. The management-invasion continuum presented here can support decision-making at multiple spatial scales, for example the entry and spread of an alien species into a new region/country, or into a defined space such as a protected area or an island. Also, it can help decision-making from a temporal dimension by identifying management actions suited for each stage of invasion. The management-invasion continuum can also be used to identify how management approaches targeting pathways, species, sites and ecosystems are inter-connected with each management objective and action.



EXAMPLES OF SUCCESS

Prevention

Zero tolerance biosecurity in Barrow Island, Australia
A risk analysis of all material and passenger pathways and a quarantine system prevent the entry of alien species to the island and surrounding marine habitats *(Box 5.2)*.

Eradication

***Ceroplastes cirripediformis* (barnacle scale)**
Eradicated by burning the infested plant stems soon after its detection in 2021 in India *(Box 5.16)*.

***Caulerpa taxifolia* (killer algae)**
A rapid response using liquid chlorine achieved the eradication in a lagoon in the United States of America *(5.5.3)*.

Containment

***Ageratina adenophora* (Croftonweed)**
National containment lines were proposed to prevent spread while a biological control program is under development in China *(5.3.1.2)*.

***Lymantria dispar* (gypsy moth)**
Using pheromone traps, spread of the species was slowed in North America *(5.5.4)*.

Control & retoration

***Sus scrofa* (feral pig)**
In the El Palmar National Park in Argentina, wild boar damaged Butia yatay (yatai palm tree). Through a controlled shooting program, abundance of wild boar was reduced to levels causing minimal damage to soil and the yatai palm tree recruitment *(Box 5.6)*.

Figure 5 1 **Conceptual diagram of the management-invasion continuum.**

The generalized invasion curve without management and the expected changes in the invasion curve with appropriate management actions in **A)** terrestrial and closed water systems (including coastal systems and salt marshes) and in **B)** marine and connected water systems. Boxes indicate the optimal management objectives at each invasion stage (**Glossary**). Colour gradient of managing

Figure 5 1

pathway, species, site and ecosystem boxes show how the relative focus generally changes as invasion progresses. Boxes below indicate typical management actions necessary to achieve each management objective. Post-establishment management actions are not shown under panel B since these are generally not achievable in these systems. In a management context, the first detection (introduction point), the lag phase and the exponential spread phases are important points to design an early detection and rapid response management plan. This figure is conceptual, and the curves do not represent actual population dynamics of invasive alien species.

For biological invasions in terrestrial and closed water ecosystems, there are generic management objectives such as prevention, early detection, eradication, containment and control and restoration associated with the status of the invasive alien species (absent, established, spread and widespread; Robertson *et al.*, 2020; **Glossary**; **Chapter 1**, **section 1.4**). Adaptive management and long-term monitoring need to be part of all modes of management (**Glossary**). Prevention is implemented by jurisdictions in the pre-entry phase and points of entry for intercepting new alien species, and by definition has to target arrivals of all alien species, not just those that may become invasive. Although context dependent, in the ensuing lag-phase (**Glossary**; **Chapter 2**, **section 2.2.1**) during establishment, opportunities may exist for eradication and the potential to flatten the invasion curve **(Figure 5.1)**. Early detection enables a rapid response to eradicate or contain an alien species before it spreads. The likelihood of eradication generally decreases during the rapid dispersal phase. Long-term species-based or site-based adaptive management approaches of invasive alien species that can no longer be eradicated or for which containment alone is not viable, can then effectively minimize biophysical and/or socioecological and socio-economic impacts.

In terrestrial and closed water ecosystems, effective management involves a series of actions, including objective decision-making (**section 5.2**), surveillance (**Glossary**) and monitoring (particularly at ports of entry; **section 5.5**) and chemical, physical and biological controls (**section 5.5.5**), all supported by a range of platforms, tools and technologies (**section 5.4**). Decision-making includes agreeing on clearly defined objectives ("why manage?") and carrying out evidence-based risk assessment and prioritization to undertake the most effective actions, responding to the questions: "what actions?", "where to take actions?" and "how to take actions?". Management programmes can focus on the following three management options, namely the pathways of introduction, the invasive alien species and the invaded sites/ecosystems singly or in combination (**section 5.3**).

Pathway, species-based and site-based strategies for the management of biological invasions are alternative or complementary approaches given particular socioecological contexts. The approach taken is dependent on the management goal, the status of invasion of an alien species along the introduction-invasion continuum **(Figure 5.1)** and the socio-economic situation. Pathway management, which aims to prevent the introduction of a species into new sites, functions across the biological invasion continuum (**Figure 5.1**; **section 5.3.1.1**), where efforts are generally aimed to prevent introduction into a new region/country, but also to manage the wider spread during the rapid expansion phase. Species-based management either proactively minimizes future impact risks (**Glossary**) through interventions to eradicate or contain new incursions of a species or targets suppression of a priority single species (e.g., through landscape level control such as classical biological control – **sections 5.4.3.2f** and **5.5.5**, or lethal control programmes – **section 5.4.3.2d**), or multiple established alien species (**Glossary**) through localized extirpation (McGeoch, Genovesi, *et al.*, 2016; Simberloff, 2013). Site-based or ecosystem-based management focuses on a specific area or ecosystem defined by its inherent value (e.g., biosphere reserves, heritage sites or protected areas; **section 5.3.2**) and the threat that invasive alien species pose to biodiversity conservation value and management objectives of that site (Owen & Sheldon, 1996). These objectives and ecosystems include the broader socioecological context (Stokols, 1996).

In marine and water connected systems, post-entry management of invasive alien species is generally ineffective (Booy *et al.*, 2020; Lehtiniemi *et al.*, 2015). Therefore, prevention at the pre-entry phase and surveillance and early detection before the establishment stage are the most effective management options in these ecosystems. In the marine context, site-based activities mainly consist of surveillance (e.g., in ports, harbours, mariculture facilities and marine protected areas). Eradication of established invasive alien species is rarely achieved in these ecosystems and may be possible only for sessile or low mobility organisms (Lehtiniemi *et al.*, 2015), with few examples of chemical control (**section 5.5.3**) in small bays or enclosed waters. Nonetheless, prevention is the optimal viable option to avoid negative consequences of invasive alien species (Galil, Danovaro, *et al.*, 2019) in marine systems given the complex nature (**Glossary**) and vastness of these environments for implementing management procedures. Early detection is important, even if eradication is not achievable, to explore the possibility of mitigation (Lehtiniemi *et al.*, 2015). In any case, all management actions would need resourcing for the costs of stakeholder engagement and communication, implementation of techniques and tools, restoration and long-term evaluation.

5.1.2 Scope of the chapter

Chapter 5, which covers all the key elements of management of biological invasions along the management-invasion continuum **(Figure 5.1)**, consists of an introduction section (**section 5.1**) and five theme-specific sections (**sections 5.2** to **5.6**), with the relationships between each shown in **Figure 5.2**. This solution-focused chapter has strong links to other chapters of the assessment, since management and decision-making are intrinsically dependent on knowledge of the impacts of invasive alien species (**Chapter 4**), their status and trends (**Chapter 2**; **Glossary**) and how to manage direct and indirect drivers of change that impact invasion (**Chapter 3**). Collective community management decisions are also supported by good understanding of Indigenous and local knowledge and how invasive alien species impact good quality of life (**Glossary**; **Chapter 4**, **sections 4.5**, **4.6**; **Chapter 1**, **section 1.6.7.1**). Lessons that can be learned from previous and current management efforts and control options presented in this chapter can inform and improve future policy options (**Glossary**; **Chapter 6**).

5.1.3 Management: challenges and opportunities

Throughout **sections 5.2** to **5.6** **(Box 5.1)**, various challenges, case studies and future opportunities have been identified and distilled as lessons to be learned.

Challenges that managers of biological invasions face are jurisdictional boundaries (Flueck, 2010; Stokes *et al.*, 2006), inadequacy of regulations (Burgiel *et al.*, 2006; Garcia-de-Lomas & Vilà, 2015), lack of expertise (Shine, 2005), poor stakeholder engagement (Driscoll *et al.*, 2014; Simberloff *et al.*, 2005; **Chapter 6**, **section 6.4**) and uncertainty on where to allocate limited resources (Prior *et al.*, 2018; **section 5.6**). Some decision makers are hesitant to attempt prevention given only a small proportion of alien species arriving may ultimately become invasive (**Chapter 1**, **Figure 1.1**), so in some cases it is perceived as best to wait until the impacts of the alien species are understood (Finnoff *et al.*, 2007; **Chapter 4**, **section 4.2**) but this can result in delays rendering subsequent management costly or even impractical. Additionally, implementing some of the management approaches may not be acceptable to all the stakeholders and Indigenous Peoples and local

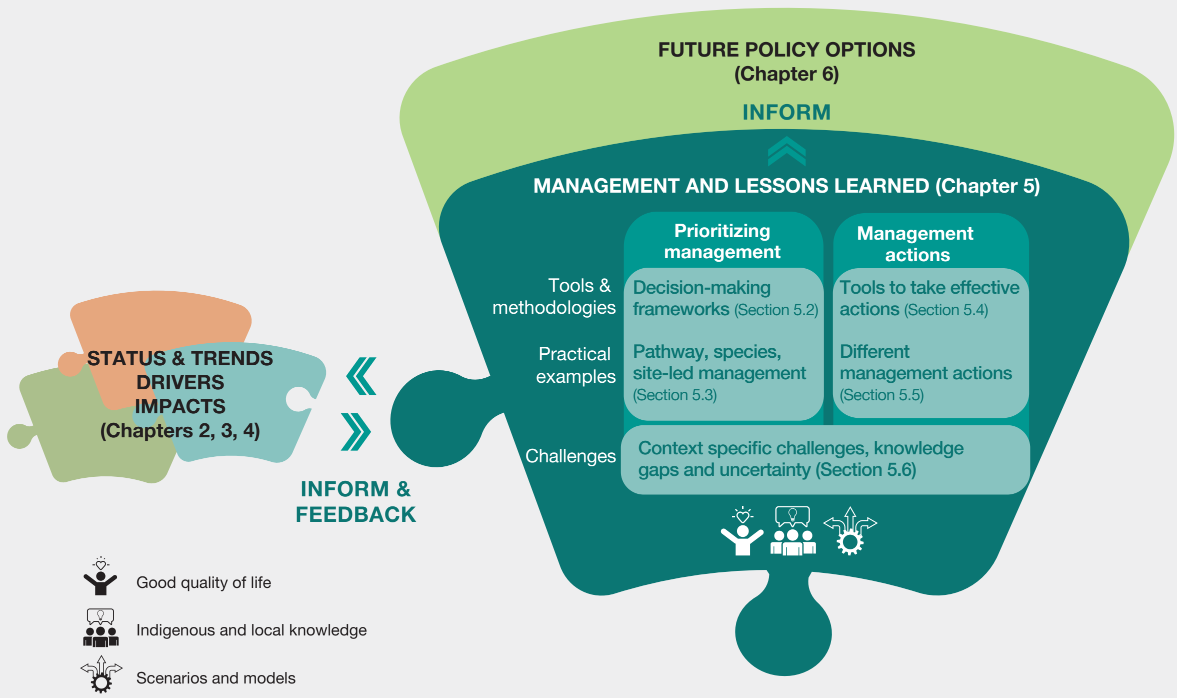


Figure 5 2 **Content of sections in this chapter and their linkages with the other chapters of the assessment.**

This figure shows the structure of **Chapter 5**, and how it connects to **Chapter 2** (status and trends of biological invasions), **Chapter 3** (direct and indirect drivers affecting biological invasions), **Chapter 4** (impacts of invasive alien species) and **Chapter 6** (future policy options to manage biological invasions). It also informs and provides feedback and clarity to all these chapters.

communities. For example, there is increasing opposition to some control methods, e.g., use of chemical pesticides or lethal control of alien vertebrates using toxins (Genovesi & Bertolino, 2001; Longcore *et al.*, 2009); concern about the non-target effects of certain methods (e.g., biological control, genetic control options), high costs of management and the paucity of funds for continuous management and monitoring (Wittenberg & Cock, 2003). Some invasive alien species are also considered by some stakeholders as beneficial (Niemiera & Holle, 2009; Scasta *et al.*, 2015; **section 5.6**; **Chapter 4**, **sections 4.3**, **4.4**, **4.5**). Removal of invasive alien species which have commercial or cultural values can deprive people who utilize these for livelihood and other adapted uses (Grechi *et al.*, 2014; N. A. Marshall *et al.*, 2011; van Wilgen, 2012), though other benefits through management of invasive alien species can also contribute to nature's contributions to people and to good quality of life. As the concept of an invasive alien species is a human construct, interests and perceptions of invasive alien species for different stakeholders and Indigenous Peoples and local communities may differ and the management of some invasive alien species may result in conflicting values (**Chapter 1**, **section 1.5.2**).

There are many opportunities for the successful prevention and management of biological invasions. Modern tools and techniques, combined with long-standing proven management methods, often involving Indigenous and local knowledge, can reduce the impacts of invasive alien species in many instances. There is an increasing global willingness and desire among stakeholders to cooperate on management, undertake collaborative research and build awareness on the impacts of invasive alien species. Promoting such collective efforts to address issues are important in the management process (**sections 5.4**, **5.5**). Scientific information and databases (**section 5.4**) and decision-making tools (**section 5.2**) are being made openly available to policymakers and resource managers to enable informed decision-making. Collaboration among governments, agricultural industries, the general public, including Indigenous Peoples and local communities, non-government institutions and land users, and concerted actions by all parties will assist in addressing the challenge in a strategic, holistic and timely manner (Reaser, 2003) at the most appropriate scales (Glen *et al.*, 2017). There is evidence to suggest that even less extensive cooperation and coordination among independent landowners can have a profound positive effect on managing invasive alien species (Epanchin-Niell & Wilen, 2015). Clear objective-driven invasive alien species management involving, and agreed upon by relevant stakeholders can bring substantial ecological and social benefits, and can eventually open new opportunities for improving good quality of life (Chikoye *et al.*, 2006; H. P. Jones *et al.*, 2016; Samways *et al.*, 2010; **Chapter 6**, **section 6.4**).

5.2 EVIDENCE BASED DECISION-MAKING

Attaining good community collaboration, co-development and governance of any form of environmental management has many challenges (Margerum & Robinson, 2016). Invasive alien species are the result of human activities, situations or events that are subject to human experiences, concerns and values (**Chapter 3**, **section 3.2.1**). Thus, the management of their threats and impacts implies effective multiway stakeholder community engagement in communication, knowledge sharing and co-development around goal setting, decision-making and intervention through action (**Chapter 6**, **section 6.4.2**). As for many environmental issues, there are also a wide variety of actions to address invasive alien species, for example, technical, legal, economic, social, behavioural, cultural or knowledge based. Whatever the action is, decisions are taken by representatives of societies, communities and individuals confronted with invasive alien species using a precautionary approach. Decision makers may be public or private, including policymakers, land and waterway managers, Indigenous Peoples and local communities, volunteers working on public land, private tenants, non-governmental organizations or community groups (IPBES, 2020). Decision-making can also be differential between communities and gender. For example, in many parts of the world women are action takers while men make most of the decisions (e.g., in smallholder crop management; Fish *et al.*, 2010; Upadhyay *et al.*, 2020). Decision-making relies on available evidence, respective values, interest and responsibilities of stakeholders, available management resources (D. L. Larson *et al.*, 2011; Piria *et al.*, 2017), and likely trade-offs.

A variety of frameworks and decision-support tools and systems have been developed to facilitate the decision-making process, linking science, policy and management (Matthies *et al.*, 2007; J. R. U. Wilson *et al.*, 2020). These can help choose between particular strategies and treatment options for management of biological invasions (Kriticos *et al.*, 2018) and can support adaptive management "learn as you go" systems (**section 5.4.3.3**). When direct management actions are required, these tools can assist in evaluating the progress or success of management (Garner & Beckett, 2005). This section describes stakeholder community engagement and knowledge-sharing frameworks, prioritization processes and available methodologies and tools for management decision-making to combat invasive alien species.

5.2.1 Stakeholder community engagement and knowledge-sharing frameworks for developing communities of practice

Approaches involving collaborative governance networks of stakeholder communities to mitigate the impacts of different invasive alien species will vary depending on context (**Chapter 1**, **section 1.5.1**; **Chapter 6**, **section 6.4.4**), with each community having different a) perspectives and engagement reasons (directly and indirectly affected communities), b) knowledge bases (understanding of drivers, processes, trends and impacts of invasive alien species) and c) roles in the response (resourcing, governing and implementing; **Chapter 1**, **section 1.5**). Effective environmental, social and cultural knowledge and governance (**Glossary**; **Chapter 6**) supports effective stakeholder community engagement and collaboration. The foundation of community engagement is building trust and understanding through knowledge sharing. For invasive alien species, this concerns knowledge of impacts on nature, good quality of life and nature's contributions to people (**Glossary**); and the likelihood that impacts can be mitigated with high benefit-cost and cost-effective management actions and that long-term system resilience (**Glossary**) benefits can result from these actions. Stakeholder engagement systems need to address the challenges of community collaboration (McAllister *et al.*, 2017), be cost-effective (S. Liu *et al.*, 2019), and grow social resilience to invasive alien species (Maclean *et al.*, 2018). Indigenous Peoples and local communities often have different motives for engagement than other stakeholders (**Supplementary material 5.1**), and manage biological invasions for multiple purposes which are closely related to each other (IPBES, 2022b). It may be noted that spirituality is an overarching motivation for Indigenous Peoples and local communities to protect their land and assets from invasive alien species, even though this is often underreported (IPBES, 2022b; **Chapter 4**, **section 4.6**). Therefore, they can provide unique knowledge and management response capacity (Bach *et al.*, 2019; Kannan *et al.*, 2016; Madegowda & Rao, 2014). These stakeholder community engagement systems can be highly context specific (e.g., low vs. high income countries, peri-urban vs. rural situations, terrestrial vs. marine environments, public vs. private, etc.) but are vital to create co-developed communities of practice around effective community-led responses that support prevention, preparedness (**Glossary**; **section 5.4.2**), rapid response and widespread control.

Collaboration and knowledge sharing among stakeholder communities (governments, scientists and non-governmental organizations) and Indigenous Peoples and local communities also help management of, or if needed adaptation to, new invasive alien species in localities and regions (IPBES, 2020). For example, the volunteering programme at the Horus Institute for Environmental Conservation and Development of Brazil with the Federal University of Santa Catarina engages university students and the local community to do hands on work controlling invasive alien pines and restoring coastal areas in the Dunas da Lagoa da Conceição Natural Municipal Park (Florianópolis, Santa Catarina state, Brazil), one of the most impacted ecosystems in the Atlantic Forest Biome in Brazil where some 470 thousand pines have been eliminated. Without the programme, invasive alien pine trees would have degraded half of the total area of the park in two decades (Dechoum *et al.*, 2019). In another example, by organizing tournaments or derbies since 2009, volunteers from Colombia, Bahamas and Florida (United States of America) helped raise awareness of *Pterois volitans* (red lionfish) and *Pterois miles* (lionfish), an Indo-Pacific invasive alien species widespread in the Caribbean region (Green *et al.*, 2017). There are two other similar initiatives described by Anderson *et al.* (2017) and Kleitou *et al.* (2021).

Community-based management of biological invasions often happens through profit-making activities such as harvesting for sale in new markets or by encouraging recreational hunters to act as management agents, however this can create conflicts (**section 5.6.1.2**). *Paralithodes camtschaticus* (red king crab) was introduced in the Barents Sea affecting local fisheries. The Saami community and other coastal fishermen communities of Norway played an important role adapting to this invasive alien species and participating in management actions with financial return and changing the fishing system (Broderstad & Eythórsson, 2014). Stakeholder communities can collectively plan options and select management interventions and evaluate and transparently communicate outcomes, recognizing potential negative and positive impacts. Societal and political support for management decision-making and engagement with Indigenous Peoples and local communities can be achieved through participatory decision-making to ensure a common understanding of the pros and cons of decisions and actions (S. Liu *et al.*, 2019). It is important that these participatory mechanisms respect social structures, intellectual property, land rights and self-determination through free, prior and informed consent, respect spiritual values and processes, including prayers, ceremonies and other ways through which relationships between humans and nature are balanced (Bajwa *et al.*, 2019; IPBES, 2020; Pretty Paint-Small, 2013). Once action has been decided, adaptive management involves observation, experimentation and collective learning to optimize outcomes (Alexander *et al.*, 2017). Communication activities imply consultation to understand and respect Indigenous Peoples and local community perspectives (IPBES, 2020). Without such engagement, conflicts may result leading potentially to loss of livelihoods, threats to cultural systems, displacement from lands, marginalization and gender inequity (e.g., as occurred with the Il Chamus in Ng'ambo pastoralists in Kenya; Mwangi & Swallow, 2005).

Standardized frameworks can support stakeholder community engagement, but still need improvement (Novoa *et al.*, 2018; R. T. Shackleton, Larson, *et al.*, 2019). Context-specific frameworks have been developed and analysed (Lansink *et al.*, 2018; McAllister *et al.*, 2015). There are a number of approaches that have been developed and applied to management of biological invasions which also support effective decision-making across multiple management options (Firn *et al.*, 2015; Carwardine *et al.*, 2019). Shackleton *et al.* (2019) argue that "to make stakeholder involvement more useful, we encourage more integrative and collaborative engagement to (1) improve co-design, co-creation and co-implementation of research and management actions; (2) promote social learning and provide feedback to stakeholders; (3) enhance collaboration and partnerships beyond the natural sciences and academia (interdisciplinary and transdisciplinary collaboration); and (4) discuss some practical and policy suggestions for improving stakeholder engagement in [biological] invasion science research and management".

5.2.2 Evidence based decision-making framework vs. *ad hoc* decision-making

How decision-making is undertaken with regards to what, where and how to manage biological invasions is rarely explicitly stated in most management contexts. Many management action decisions are done in an *ad hoc* way as a flexible emergency response to new incursions, a belated observed impact, or as a political imperative (Sheail, 2003). Often decisions on actions need to be made under a degree of uncertainty, such as when containing new incursion to avoid spread. Sometimes the science lags behind the operational tools required. Less often is there a formal community or government framework in place for decision-making.

Preparedness is improved through adopting a systems-based adaptive management approach allowing learning to lead to improvements. Explicitly addressing the rationale behind decision-making allows for transparency, repeatability, learning as well as endorsement and support for the actions resulting from the decision-making process (De Fine Licht, 2014; Estévez *et al.*, 2013; Moon *et al.*, 2015, 2017; Vanderhoeven *et al.*, 2017). This practice is the basis of some international standards such as the International Standards for Phytosanitary Measures (ISPM) of the International Plant Protection Convention (IPPC; IPPC, 2019; **Supplementary material 5.8**) and World Organisation for Animal Health (WOAH, founded as OIE) standards (World Organisation for Animal Health, 2020).

Conceptually, actions are often categorized into three main approaches **(Figure 5.3)** all of which are underpinned

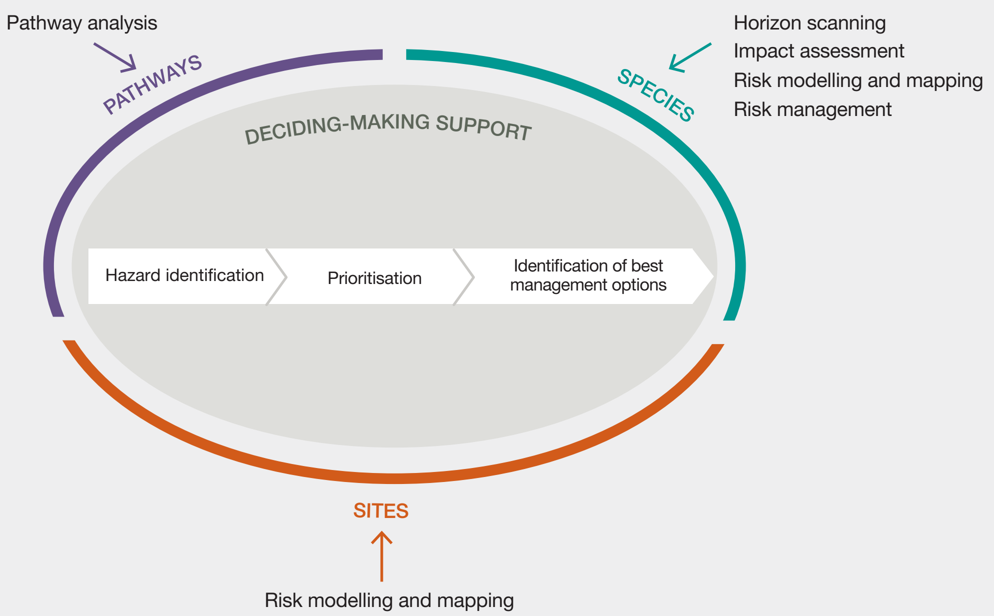


Figure 5 3 **The three main approaches to support decision-making actions on management of biological invasions and examples of contributing tools.**

Decision-making relies on a sequence of steps consisting of the hazard identification, the prioritization of threats and the identification of the best management options. This applies whether the decision deals with pathways, species, or sites.

by decision-making to identify and prioritize targets and management actions:

- Focusing on the pathways of introduction of alien/invasive alien species: this approach aims to answer questions such as "Is pathway 'X' a major source of supply of alien species for a given area?"; "Which are the most high-risk pathways for the arrival of invasive alien species?"; and "What is likely the most effective way to reduce the introduction of species *via* this pathway?"

- Focusing on the alien and/or invasive alien species of interest: this approach aims to answer questions such as "Is species 'X' impacting a given region?"; "Which species is most at risk of entry and establishment in a given jurisdiction?"; "Which species is most impactful?"; "How established or spread is this species?"; "Can it be contained or eradicated?"; "What is likely to be the most effective management action?"; or "How much will it cost?"

- Focusing on invaded sites: this approach aims to answer questions such as "What sites harbour the most sensitive habitats prone to invasive alien species establishment?"; "Does this site similarly include site Y which provides important ecosystem services to livelihoods?"; "Which sites are a high priority for management based on the level of invasion or the value of the biodiversity and ecological assets at that site?"; and "What are the best actions to manage high priority sites?"

A "site" here is a clearly defined land/freshwater/marine area including the sociological system (i.e., the social, institutional and cultural contexts of the relationship between people and the environment; Stokols, 1996). Management of biological invasions is also considered holistically in decision-making frameworks for ecosystem-based management, increasingly applied (Espinosa-Romero *et al.*, 2011; Lampert *et al.*, 2014; Tanentzap *et al.*, 2009).

5.2.2.1 Analytical approaches, methods and tools for decision-making support

Many analytical approaches, methods and tools are available to address the types of decisions under species, pathways and site management approaches, aimed at mobilizing existing knowledge around the three analytical elements. These are hazard identification, prioritization and

Table 5 1 **Analytical approaches, tools and methods available to making decisions about biological invasions.**

This table shows the different analytical approaches, tools and methods available to decision makers to tackle biological invasions as well as their respective utility and different levels of governance to which they apply.

Tools and Methods	Utility		
	Hazard identification	Prioritization	Identification of best management options
Horizon scanning	X	X	
Pathway analysis	X	X	
Impact assessment		X	
Risk analysis	X	X	X
Risk assessment	X	X	
Risk modelling and mapping	X	X	
Risk management		X	X
Economic approaches		X	X
Multi-criteria analyses	X	X	X
Case study learnings from past successes/failures			X
Evidence synthesis			X
Best management practices			X

identification of best management options (**Figure 5.3**; **Table 5.1**). Approaches, methods and tools may be used in isolation or in a complementary manner for answering common questions across the different approaches. The analytical approaches, methods and tools presented below align with the conceptual biological invasion process (**Chapter 1**, **Figure 1.6**; **Figure 5.4**). Several tools rely on information in previous chapters, including up to date knowledge of species distribution and abundance (**Chapter 2**), direct and indirect drivers facilitating biological invasions across the invasion continuum (**Chapter 3**) and invasive alien species impacts (**Chapter 4**). This highlights the importance of knowledge and data for evidence-based decision-making on management.

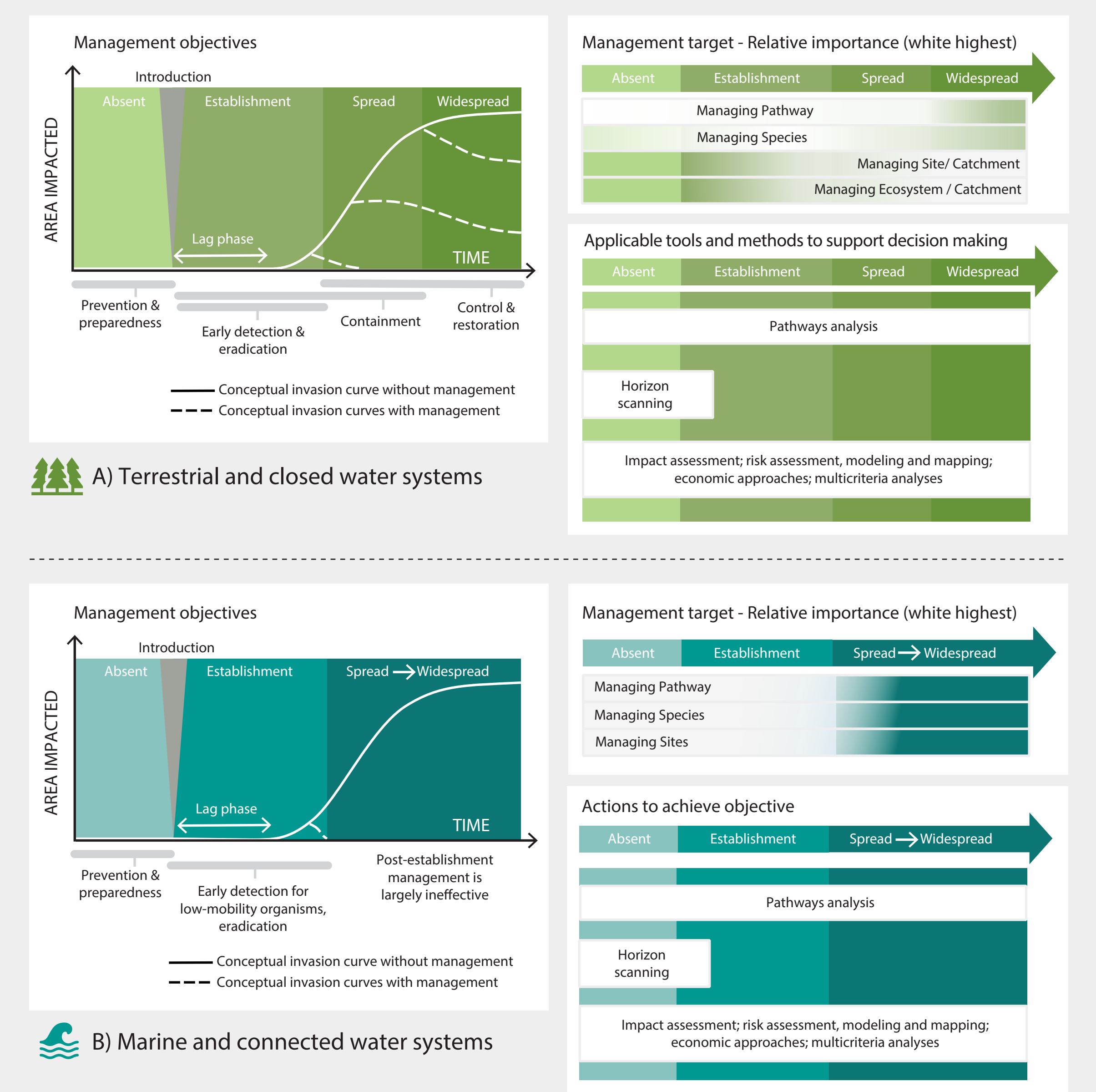


Figure 5 4 **Applicability of different tools and methods along the conceptual diagram of management along the invasion curve which provides a continuum for management interventions.**

This shows how tools and methods support decision-making in relation to the management targets for A) terrestrial and closed water systems (including coastal systems and salt marshes) and B) in marine and connected water systems. Gradients indicate the management target and the associated tools and methods necessary to support management decision-making, as well as their relative focus changing as invasion progresses. This figure is conceptual, and the curves do not represent actual population dynamics of invasive alien species.

The range of tools presented below is not intended to be exhaustive but to illustrate the diversity of tools available to meet different decision-making objectives.

a) Horizon scanning

Horizon scanning is the systematic examination of emerging and future potential threats and opportunities within a given context (Food Standard Agency, 2018) and has been used to prioritize potentially new alien species threats in jurisdictions supporting prevention and preparedness (Copp *et al.*, 2007; H. E. Roy *et al.*, 2014). Horizon scanning has been considered for discrete taxonomic groups, such as plants (Andreu & Vilà, 2010) or animals (Parrot *et al.*, 2009), and specific environments such as freshwater (Gallardo & Aldridge, 2013), at the national level (Lucy *et al.*, 2020; Peyton *et al.*, 2019) or for a wider region (H. E. Roy, Bacher, *et al.*, 2018) and globally (Dawson *et al.*, 2022). Horizon scanning usually follows a structured process based on some form of impact or risk assessment, often involving expert elicitation, to reduce and simplify a long list of potential invasive alien species to a prioritized subset. The inherent lack of evidence for horizon scanning, compared to risk assessment of established species, results in uncertainty but this can be documented through the process (H. E. Roy *et al.*, 2020). This has been applied in the United Kingdom (H. E. Roy *et al.*, 2014), where predictions subsequently supported future arrivals of eight out of the top ten species, including *Dreissena rostriformis bugensis* (quagga mussel; Aldridge *et al.*, 2014) and *Vespa velutina* (Asian hornet; Keeling *et al.*, 2017). The Centre for Agriculture and Bioscience International (CABI) has developed an online Horizon Scanning Tool quickly allowing identification and categorization of species that might enter one geographic area from another (CABI, 2021). Improvements to the process to quantify the likelihoods of economic, environmental and social impacts of species with no prior history of introduction outside their native range (**Glossary**) are now starting to be included (Peyton *et al.*, 2019, 2020).

b) Pathway risk analysis

Introduction pathway (**Glossary**) risk assessment, supported by standardized pathway categorization (IUCN, 2017), is needed to support pathway management decision-making, regardless of the geographical context or the many potential taxon–pathway combinations (A. P. Robinson *et al.*, 2017). The use of standard pathway categories allows to readily collate and compare introductions to prioritize pathways (Faulkner *et al.*, 2020; McGrannachan *et al.*, 2021; Saul *et al.*, 2017).

Analysis and prioritization of pathways supports regulatory approaches likely to also use and compare pathway data on commodities and vectors (McGeoch, Genovesi, *et al.*, 2016; **Chapter 3**, **section 3.1.1**; **Glossary**). Quantitative pathway risk analysis requires a set of key variables (Essl *et al.*, 2015; Hulme, 2009):

- Historical strength of the association between the species threat and the commodity, vector or pathway at the point of departure;
- Origin, volume and type of commodities or vector introduced for each pathway;
- Frequency of introduction;
- Species survivorship and population growth during transport/storage;
- Environmental suitability in the region of introduction for species establishment (e.g., climate matching);
- Time of year relevant for species establishment following introduction;
- Ease of species detection and identification;
- Effectiveness of management measures;
- Movement of the commodity or vector in the region following introduction;
- Likelihood of transfer from port of entry to a suitable habitat.

As such parameters are only known for very few alien species and only for specific pathways, pathway risk analysis (and management) remains challenging, requiring inferences based on statistical aggregates across species, but have been undertaken in different contexts (Leung *et al.*, 2014; Nunes *et al.*, 2015). Costello *et al.* (2007) developed an analytical model linking alien species introductions to trade volumes. In practice, pathway risk analyses often rely on information on the range of vectors and routes by which alien propagules are introduced, and their respective propagule loads (McGeoch, Genovesi, *et al.*, 2016). For example, a pathway risk analysis performed for the Antarctic identified high propagule loads linked with the importation of fresh produce (Hughes *et al.*, 2011), infrastructure development activities, and entrainment on the clothing of visiting tourists and scientists (Chown *et al.*, 2012). A similar pathway analysis for Barrow Island introductions provided details on the type of organism detected (classification and state; e.g., seed) attributed to specific pathways at and post-border (**Box 5.2** in **section 5.3.1.1**; Scott *et al.*, 2017).

The temporally dynamic nature of the introduction pathways for alien species makes pathway risk analysis particularly difficult to perform (Piel *et al.*, 2007), but is helped by increasing, harmonizing and consolidating pathway information across multiple sources and explicitly using border interception (Trouvé & Robinson, 2021) and post-border detection data (Essl *et al.*, 2015; Saul *et al.*, 2017) to strengthen pathway risk analyses. Postal mail is an explicit pathway risk for invasive alien species and understanding how mail inspections avoid biosecurity risks helps manage this pathway (S. Clarke *et al.*, 2018). Most attempts to model pathways have focused on describing the likelihood of species introduction and establishment (Bradie & Leung, 2015; Paini *et al.*, 2016) and rather few have attempted to address explicit strategies for the management of biological invasions(Hulme, 2009). García-Diaz *et al.* (2017) demonstrated that biosecurity activities implemented in Australia have decreased introduction probability of alien amphibian stowaways, in turn reducing the likelihood of a virus-infected animal entering the country. For pathway risk analysis the number of amphibian interceptions across six Australian States were more positively related to the amount of shipping than air transport. Risk assessment has also been recently applied to understand the pathways of introduction for marine invasive alien species incursions (K. R. Hayes *et al.*, 2019) and guidelines on pathways risk analysis application have been developed for the attention of sectors such as aquaculture (FAO, 2008).

c) Species impact assessment

Understanding and predicting the magnitude of actual or potential impacts of invasive alien species is key to deciding whether management actions are required (**Chapter 4**, **section 4.7**). Alien species impact assessments often differ in purpose, taxonomic scope, spatial scale and methods; often with bespoke ways of characterizing and assessing uncertainty. Some consider only environmental impacts (Van der Colff *et al.*, 2020; Vanderhoeven *et al.*, 2015) whereas others also include socio-economic impacts (Bacher *et al.*, 2018; **Chapter 4**, **Box 4.13**). Some protocols were designed to be taxonomically generic (Sandvik *et al.*, 2019) whereas others were developed for specific environments (Olenin *et al.*, 2007) or specific taxonomic groups. Some evaluation procedures rely on a panel of assessors to participate in the assessment (Kumschick, Bacher, *et al.*, 2020; Volery *et al.*, 2020), others are based on assessments performed by single experts or do not clarify this (Vanderhoeven *et al.*, 2017). Impact assessment systems have also been standardized, such as the Environmental Impact Classification for Alien Taxa (EICAT; IUCN, 2020b) and the Socio-Economic Impact Classification of Alien Taxa (SEICAT; Bacher *et al.*, 2018; **Chapter 4**). EICAT has been developed by the International Union for Conservation of Nature (IUCN; IUCN, 2020b; Van der Colff *et al.*, 2020). Unlike risk assessment, impact assessment does not consider the likelihood of establishment or spread following introduction.

d) Risk analysis

Risk analysis is a process of three complementary components: 1) risk assessment, supported by risk modelling and mapping; 2) risk management and 3) risk communication (EFSA Scientific Committee & Scientific Opinion on Risk Assessment Terminology, 2012; Geering & Lubroth, 2002; IPPC, 2019; Lanzoni *et al.*, 2019). These components are often undertaken independently and are described hereafter. Risk communication is often not explicit or even absent from decision-making processes.

As expressed by Liu *et al.* (2011), "the separation of risk assessment and management disrupts essential connections between the social values at stake in risk management and the scientific research involved in gauging the likely impacts of management actions, leaving the (…) decisions to be made in the wake of political pressures that reflect competing views on the proper trade-offs among competing values."

In order to improve the reliability of expert-based risk analysis, Vanderhoeven *et al.* (2017) provided the following eight recommendations:

1. Clearly define the scope and objective of any risk analysis;
2. Select an appropriate risk analysis/assessment approach;
3. Gather all baseline data and available information;
4. Identify missing data and information;
5. Define clear and transparent quality control procedures such as a peer-reviewing or consensus building;
6. Explicitly address feasibility of management;
7. Explicitly consider uncertainty in the analysis (assess level of confidence, quantify level of agreement among experts and highlight context-dependent variability); and
8. Explicitly consider uncertainty in risk communication.

Risk analysis requires mechanisms for acquiring expert information and opinion through an unbiased expert elicitation process (Burgman, 2005).

Detailed risk analyses are based on probability analyses and are complicated quantitative expert elicitation processes based on Bayesian belief networks and probability distributions. Although they have been applied in some invasion contexts such as the proposed release of genetically modified organisms (K. R. Hayes, Hosack, Dana, *et al.*, 2018), they are generally too costly for the assessment of most invasive alien species risks.

e) Risk assessment

The notion of "risk" is the chance that a particular hazardous event may actually cause harm, and is regarded as a product of three factors: *exposure x likelihood x consequence* (Kinney & Wiruth, 1976; A. P. Robinson *et al.*, 2017). For invasive alien species, *exposure* results from the successive introductions, establishments and spread of an alien organism, whereas *likelihood* and *consequence* underpin the impact assessment referred to in section c (D'hondt *et al.*, 2015). *Likelihood* is the probability that an invasive alien species impact affects nature, nature's contributions to people and good quality of life; and *consequence* is the magnitude of impact if it occurs.

Most commonly risk assessment is a relatively straightforward semi-quantitative approach based on a scoring system where different components of risk are assessed and scored, and a total score is obtained in some manner to define the overall level of risk. This was initially developed for assessing import risks of alien plants ("Weed Risk Assessment"; Bomford & Hart, 1999; Pheloung *et al.*, 1999). This approach is relatively quick and incurs only low cost per species and therefore is most favored by policy makers (e.g., Bomford & Hart, 1999; Pheloung *et al.*, 1999) and such approaches have been adopted as international standards (Devorshak, 2012; IPPC, 2019). Risk assessment tools have been established for particular types of invasive alien species in different parts of the world (Essl *et al.*, 2011; Groves *et al.*, 2001), including Australia (Pheloung *et al.*, 1999; Scott & Panetta, 1993), North America (Hiebert & Stubbendieck, 1993; Kolar & Lodge, 2002; Reichard & Hamilton, 1997), South Africa (Tucker & Richardson, 1995), Brazil (Ziller *et al.*, 2019) and Japan (Nishida *et al.*, 2009). In Europe a diversity of risk assessment systems have been developed (Baker *et al.*, 2008; Copp *et al.*, 2005; D'hondt *et al.*, 2015; Essl *et al.*, 2011; Gollasch & Nehring, 2006), including for specific use in sectorial activities such as aquaculture (Copp *et al.*, 2016). Each system has its own characteristics and decision-making contexts including taxonomic focus, geographical scope, type of environment, type of impact considered, scoring of impact, uncertainty consideration and expert contribution to the assessment (H. E. Roy, Rabitsch, *et al.*, 2018). Nonetheless, all systems follow the three factor standard premise and synthesize information based on formalized criteria to determine the overall risk. While risk assessment approaches based on scoring systems are relatively quick and easy, their effectiveness is rarely evaluated and there remain many shortcomings (Hulme, 2012).

Some initiatives have established repositories or databases giving access to available species risk assessments existing for a given territory, like the Canadian Invasive Species Center[2] or region, like the European Commission (CIRCABC, 2021). A comprehensive review of more than 1,000 risk assessment results is available for species that are invasive in Brazil from the National Invasive Species Database.[3] The database is open access and available in English, Portuguese and Spanish. Such an initiative does not currently exist on a global scale.

f) Risk modelling and mapping

Risk assessments are often supported by projection models to help evaluate species pathways, entry, establishment, spread and/or impact within an area of interest (Beaumont *et al.*, 2009, 2014; Elith, 2017; A. P. Robinson *et al.*, 2017; Stevenson, 2004; Venette *et al.*, 2010). Spatially modelling potential alien species distribution is a common practice for species either not present or of limited distribution (**Chapter 1**, **section 1.6.7.3**). Simple models generally based on species distribution data and climate matching software are often crude and can exaggerate the risks. Considering additional environmental data is one way to fine tune spread and distribution. Such models can also project likely future distributions under climate change (Kriticos *et al.*, 2005; Venette *et al.*, 2010). More complex process-based models can incorporate physiological limits of the target invasive alien species to better define habitat suitability (Kriticos *et al.*, 2020). Similar approaches have been applied for predicting and mapping habitat suitability and distribution of invasive alien vertebrate species, invasive alien arthropods, invasive alien plant pests and pathogens and biological control agents released to manage invasive alien plants (Haye *et al.*, 2018; Kriticos *et al.*, 2009, 2013). The maps generated can be used to guide decisions regarding the implementation of geographically targeted monitoring which allows early warning and rapid response (T. P. Robinson *et al.*, 2010). Creating accurate risk maps relies on available spatial data of species distribution, areas of interest, and climate, species physiological tolerances and environmental data layers. This is nowadays facilitated by geographic information systems (GIS) and facilities such as Global Biodiversity Information Facility (GBIF), having the capacity to process and give access to spatial data sets worldwide (McGeoch, Groom, *et al.*, 2016). Models are selected, calibrated, and verified to satisfy underlying assumptions and validated where possible against independent data. Results are depicted on maps and interpreted relative to uncertainty in the models (Yemshanov *et al.*, 2015).

Although these methods are not unchallenged, and despite a continuing debate on the accuracy of the different modelling algorithms, on the relevance and reliability of environmental data sources and on transferability of models (Capinha *et al.*, 2018; Datta *et al.*, 2020; Liu *et al.*, 2020), methodologies are still improving (Chapman *et al.*, 2019; Mainali *et al.*, 2015). Maps are commonly produced to

2. https://www.invasivespeciescentre.ca/invasive-species/what-is-at-risk/invasive-species-risk-assessment/

3. http://bd.institutohorus.org.br

illustrate potential risks from invasive alien species under different climate change scenarios (Venette, 2015; Venette *et al.*, 2010; **sections 5.6.1.3**, **5.6.3.2**), and validated in their successful prediction of invasive alien species establishment and spread (Barbet-Massin *et al.*, 2018). For example, the Tool for Assessing Pest and Pathogen Aerial Spread (TAPPAS) is an online platform for modelling the dispersal and impact of pests and diseases (Durr *et al.*, 2017). It can be used to assess the likelihood that a given pest or pathogen will be wind transported from a location where it is established, using global air current data in support of ongoing eradication or control programmes. Non-expert users can run climate-based scenarios for the spread of a given pest or pathogen in near-real-time, with downloadable data and visualization of results as risk-maps. Baysian regression models are also being used to create frameworks that can anticipate the likelihood of illegal wildlife trade of particular pet species of global popularity through wildlife smuggling pathways (Stringham *et al.*, 2021).

g) Risk management

Risk management is an extension of risk assessment to help prioritize species for management. It evaluates the implementation (feasibility and likelihood of success) of management options to reduce the known risks from invasive alien species (FAO, 1995; A. P. Robinson *et al.*, 2017). Very few risk management schemes specifically dedicated to invasive alien species exist. However, elaborate taxonomic or sector specific schemes are available for invasive alien plants (Auld & Johnson, 2014; Downey *et al.*, 2010), plant health (EFSA Panel on Plant Health (PLH), 2010) and the release of alien organisms as biological control agents (A. W. Sheppard *et al.*, 2003; van Klinken *et al.*, 2016). Governments are adopting risk management systems for some groups of invasive alien species to assist decision-making (e.g., in Australia; Department of Primary Industries, New South Wales, 2017; IUCN, 2020a). The “Non-Native Risk management scheme” is a protocol developed in the United Kingdom to assess a wide range of taxa from different environments and compare them directly according to the overall feasibility of eradication (Booy *et al.*, 2017). The management objective is first defined based on semi-quantitative response and associated confidence scores to evaluate key criteria: effectiveness, practicality, cost, impact of management, acceptability, window of opportunity and likelihood of re-invasion. Scores are obtained using expert judgement, based on available evidence, supported by consensus-building methods. This scheme has been used at different geographical scales (Booy *et al.*, 2020; Osunkoya *et al.*, 2019) and adapted in Belgium to evaluate alternative management strategies to eradication, in particular “spread limitation” (containment) and to support national management objectives (Adriaens *et al.*, 2019). Risk management systems and processes are also used to manage importation pathways to minimize the risk of alien species importations (van Klinken *et al.*, 2020).

h) Risk communication

Risk communication is the interactive process of exchange of information and opinions among individuals, groups and institutions concerning a risk or potential risk (A. P. Robinson *et al.*, 2017; Lundgren & McMakin, 2018). Communicating risks is often challenging given inherent levels of uncertainty. Ignoring uncertainty results in over-confident decisions or exaggerating uncertainty can lead to inaction in the face of mounting impacts (S. Liu *et al.*, 2011; McGeoch *et al.*, 2012). High stakeholder engagement throughout the risk analysis process, if transparent, promotes trust and confidence in decision makers (Estévez *et al.*, 2015; Groom, Strubbe, *et al.*, 2019; van der Bles, 2019; Vanderhoeven *et al.*, 2017; e.g., the TrIAScycle, **Figure 5.5**).

i) Cost-benefit, cost-effectiveness analyses and other economic approaches

Economic analysis for biological invasions generally consists of a) the actual or potential economic damages caused and b) costs of one or multiple management options optimized to minimize the combined impact and management costs (Hoagland & Jin, 2006; **Chapter 4**, **Box 4.12**). Cost-benefit analysis (**Glossary**) has been a standard approach for over 100 years and is widely applied to support decision-making for management of biological invasions (Courtois *et al.*, 2018; Naidoo & Ricketts, 2006; A. P. Robinson *et al.*, 2017) to generate a benefit-cost ratio (Naidoo & Ricketts, 2006). Socioecological systems generate inherent challenges and uncertainties associated with these approaches. When economic data are poor or lacking (Donlan & Wilcox, 2007) inclusion of even broad cost and benefit estimates have proven valuable for deciding conservation actions (Boyd *et al.*, 2015). Cost-benefit analysis can also inform the appropriate choice of biosecurity interventions across pathway, species-based (**Chapter 4**, **section 4.2**), or site-based, management responses. Portfolio theory-based decision-making or return on investment analysis (Boyd *et al.*, 2015) is another approach seeking the strategy with the best return on investment while taking into account uncertainties (Akter *et al.*, 2015; Finnoff *et al.*, 2007). Economic analysis can also support and supplement risk management where the costs of management are a component of this (Fernandes *et al.*, 2016). Economic analysis is less relevant for understanding impacts to environmental assets as the costs are hard to estimate (i.e., are intangible costs). Studies that attempt to put a value on ecosystem services are one way to address this, but generally in these contexts other approaches are used which are collectively termed cost-effectiveness analysis (**Glossary**). This aims to identify the most cost-effective management option to achieve a particular desirable

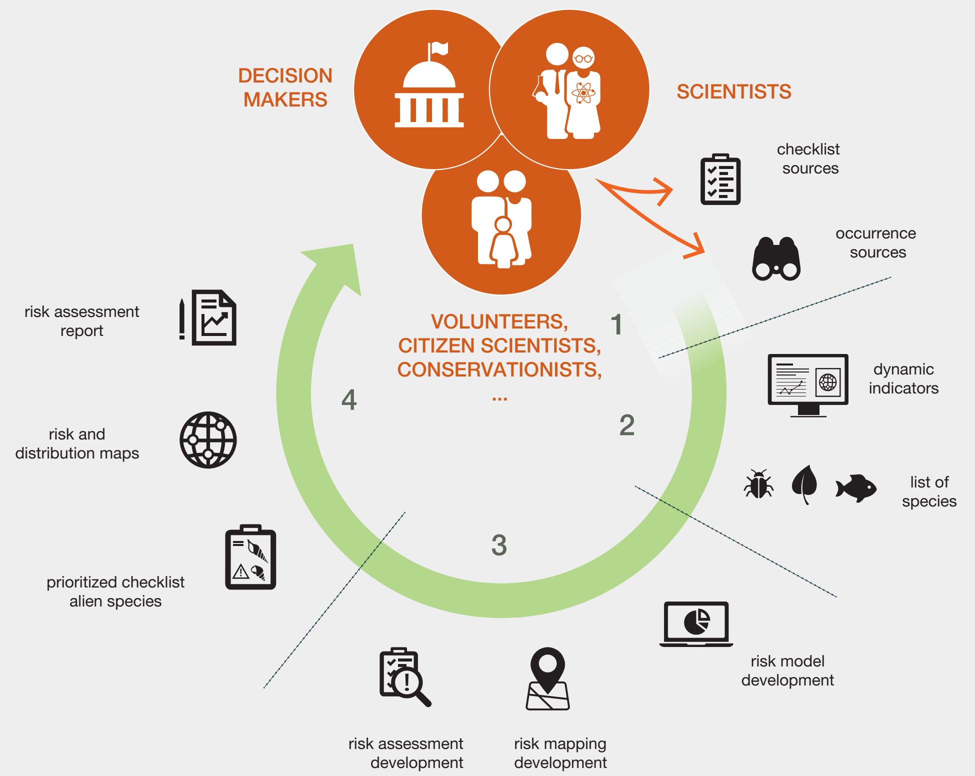


Figure 5 5 **The TrIAScycle – an example of knowledge flow allowing for transparency and promoting trust and confidence between stakeholders and decision makers.**

In this workflow, data and knowledge generated by all stakeholders (visualized in the circles at the top, i.e., scientists, public authorities, field managers and citizens involved in citizen science initiatives) are used to build a series of databases, indicators, risk assessments and communication tools to inform these stakeholders and take informed action. Source: Groom, Strubbe, *et al.* (2019), https://doi.org/10.5334/cstp.238, under license CC BY 4.0.

outcome (Drechsler *et al.*, 2016; Laycock *et al.*, 2009). Cost-effectiveness analysis generally requires unbiased expert-elicitation using a recognized transparent and documentable process. All such analyses can include economic, biodiversity, environmental, social and good quality of life considerations (Bithas *et al.*, 2018; IUCN, 2018; Rai & Scarborough, 2013). These can include tangible costs, such as costs of removing an invasive alien species from a particular location, or intangible costs or impacts such as lost biodiversity. Intangible costs can be estimated using approximations and mathematical simulations (Leung Brian *et al.*, 2002), or through monetarization approaches such as hedonic pricing (Horsch & Lewis, 2009), the travel-cost approach (Du Preez *et al.*, 2012) or contingent valuation where the stakeholder's willingness to pay (**Glossary**) for the invasive alien species impacts such as a lost ecosystem service is used to balance the benefits against the management costs (B. Provencher *et al.*, 2012). The IPBES Methodological Assessment of the Diverse Values and Valuation of Nature (IPBES, 2022a) presents diverse valuation methodologies and approaches that acknowledge, bridge and integrate the diverse values and valuation methodologies for policy and decision-making support.

j) Multi-criteria analyses

Decision-making for management of biological invasions frequently involves trade-offs between complex and conflicting environmental, social and economic objectives, potentially resulting in positive or negative consequences for different stakeholder groups (R. Gregory *et al.*, 2006).

Multi-criteria analysis evaluates multiple objectives against multiple criteria that represent competing values (Lahdelma *et al.*, 2000) and is sometimes coupled with species distribution modelling (T. P. Robinson *et al.*, 2010; **Chapter 1**, **section 1.6.7.3**). Expert elicitation is also used when available information is incomplete or imprecise. A multi-criteria analysis approach is often used to support or conducted in unison with risk management, impact or risk assessments and has proven useful for evaluating alien species threats and impacts and deciding on management options (D. Cook & Proctor, 2007; G. G. Forsyth *et al.*, 2012; Monterroso *et al.*, 2011). A simple form of multi-criteria analysis is risk-cost-benefit analysis applied to the selection of biocontrol agents proposed for the management of invasive alien plants, where all potential agent risks and benefits are identified followed by exposure analysis (likelihood of each risk or benefit occurring and the likely magnitude of the impact should it occur; Sheppard *et al.*, 2003). Deliberative multi-criteria analysis combined with participatory stakeholder engagement (Proctor & Drechsler, 2006) and facilitates consensus building and social learning (S. Liu *et al.*, 2010, 2011) and can take account of trade-offs. For example, in a case study in Western Australia, a jury was asked to prioritize a set of plant pests and diseases with different agricultural, social or environmental impacts (D. Cook & Proctor, 2007) when provided with relevant biological, ecological and economic knowledge based on perceived significance to the State's biosecurity system. The recommendations from the evaluation contrasted with the allocation of resources to the management of these species at the time.

k) Documenting successes and failures in biological invasions decision-making

Reviews of reports of successful or failed management actions, approaches or programmes available in peer-reviewed scientific studies, databases, books and published and unpublished reports across all taxonomic groups, environments or geographical areas can be used to inform future decisions for the management of biological invasions. Sutherland (2022) has promoted the value of this approach for biodiversity conservation more generally. Such information sources can inform and potentially inspire managers confronted with the same invasive alien species or a similar environmental context. Such repositories are, however, rarely developed, compiled or presented to support decision-making (Matzek *et al.*, 2014; McNie, 2007). Duplicating a successful approach often seems to be an easy decision but each context will have specific differences and challenges, so compiling many similar case studies can assist understanding when a management decision is more likely to be successful across multiple contexts.

l) Evidence synthesis

Evidence synthesis compiles individual studies within the context of global knowledge on a specific issue. It is often the basis of both evidence-based policy and practice (Dicks *et al.*, 2014). The resulting syntheses can provide rigorous knowledge for translating research into decision-support. Evidence synthesis requires an explicit and transparent question-based methodology targeting the identification, selection, appraisal and analysis of evidence from all available studies. The advantage of this approach is that all studies in a given context are assessed collectively. An example is the *Conservation Evidence* initiative, which is a collated free authoritative information resource designed to support decision-making to maintain and restore global biodiversity (Sutherland *et al.*, 2019). In 2017, the synopsis of this initiative published in print and as an open access online resource is a directory of 161 evidence-based interventions for managing freshwater taxa which were considered of high risk to Great Britain's ecosystems or economy (Aldridge *et al.*, 2015).[4] Of the 161 actions identified in this particular case, 62 per cent were not tested in any study, 20 per cent were considered "likely beneficial", 8 per cent were "unlikely to be beneficial", 5 per cent were "beneficial", 4 per cent showed "unknown effectiveness", and four studies reported "trade-off between benefit and harms" for one single action. While evidence synthesis has not yet been applied explicitly globally, it has been applied in some contexts and it is widely recognized that this approach could provide significant benefit for management of biological invasions (P. A. Martin *et al.*, 2020).

m) Best management practice approach

The best management practice approach brings together techniques and methods that have proven most effective. Such information is generally compiled through a context-specific evidence synthesis approach into a guide for addressing the management of an individual invasive alien species or a set of species generally in a particular ecological or biogeographic context. For example, the Prefectura Naval Argentina has a best practice manual for cleaning and maintaining maritime infrastructure to avoid dispersal and new introductions of marine invasive alien species (Argentine Naval Prefecture, 2021). As with case studies and evidence synthesis, similar management efforts are compared and analysed from which best practice collectively emerges. Guidelines exist for developing best practice for the management of biological invasions, including preventive strategies, eradication, containment and control (Adriaens *et al.*, 2018). Best practice management guides are being developed around the world and are generally designed for use within particular jurisdictions, based on local regulatory contexts (e.g., for use of chemicals) for management. Best practice guides can also target specific audiences (government agencies, hunters, anglers, reserve managers, or the general public). For example, the Invasive Species Council of Ontario (Canada) has developed 15 best management practice guides for invasive alien plants that also provide a historical background and taxonomic characteristics of each species (Ontario Invasive Plant Council, 2021). The series promotes the use of integrated pest management (**Glossary**) to achieve effective control and is updated on a regular basis.

4. https://www.conservationevidence.com/

5.2.2.2 How to prioritize management actions?

Management decision-making usually requires some form of prioritization. Management prioritization, whether it be in the context of pathways, species-based and site- or ecosystem-based approaches, often combines approaches, tools and methodologies presented in the previous section **(Figure 5.6)**. Species prioritization activities are the most common (Heikkilä, 2011) including being used early on in the analytical process to select species for risk assessment to optimize limited resources (Brunel *et al.*, 2010). Holistic, transparent and easy to use prioritization frameworks exist to help allocate limited resources to management actions which are expected to provide the greatest environmental and societal benefits (Bottrill *et al.*, 2008; K. A. Wilson *et al.*, 2007). Such frameworks can ensure prioritization replicable in different geographical or temporal contexts (Heikkilä, 2011). Prioritization ensures efficient resource allocation, increased transparency in collective decision-making (Kumschick *et al.*, 2012) and quantitative support to decision-making when there are conflicting objectives or measurable outcomes (Heikkilä, 2011). Prioritization for management of biological invasions is generally undertaken by public organizations, where scientific evidence may only be part of the decision-making process. Lobbying, public opinion and politics also influence prioritized decision-making around ranking invasive alien species for management. Aichi Target 9 of the Strategic Plan for Biodiversity 2011-2020 included prioritization of pathways and invasive alien species (UNEP, 2011), however McGeoch, Genovesi, *et al.* (2016) argued that any comprehensive and strategic approach to priority setting should include prioritization of pathways, species and sites. Prioritization can support prevention and preparedness along the invasion curve by ranking the high-risk introduction pathways for particular invasive alien species through to determining which sites are at greatest risk of invasion to help optimize surveillance. Prioritization can also help site selection for containment and management of established widespread invasive alien species.

Spatially explicit prioritization for the establishment of management strategies has also been proposed for example by Januchowski-Hartley *et al.* (2011) aiming at minimizing costs and the likelihood of reinvasion using the invasive tropical macrophyte *Hymenachne amplexicaulis* (hymenachne) affecting freshwater water quality, biodiversity and fisheries as a case study.

Defining overarching management objectives is important before undertaking prioritization-based invasive alien species

Figure 5 6 **Pathways, species and sites: three foci for a comprehensive approach to prioritization of management actions in biological invasions.**

This approach follows McGeoch *et al.* (2016). Combining these foci allows to 1. prioritize sites exposed to species introduced through priority pathways; 2. prioritize high-risk species introduced through priority pathways; 3. prioritize sites exposed to a set of high-risk species; 4. prioritize sites exposed to a set of high-risk species introduced through priority pathways.

management decision-making (**Box 5.10** in **section 5.3.3**). Below are some case studies.

a) Pathway prioritization, a case study from Great Britain (United Kingdom)

A method for prioritizing pathways was developed in Great Britain using established species that arrived *via* different pathways (following the classification of the Convention on Biological Diversity (CBD; CBD, 2014) and their impacts (Booy, 2019; DEFRA, 2019)). An existing dataset of the negative impacts on biodiversity of all established alien species was rated on a five-point semi- quantitative logarithmic scale (minimal = 0.0001, massive = 1) using criteria adapted from the EICAT (**Chapter 4**; Volery *et al.*, 2020b). The sum of impact scores for species introduced by

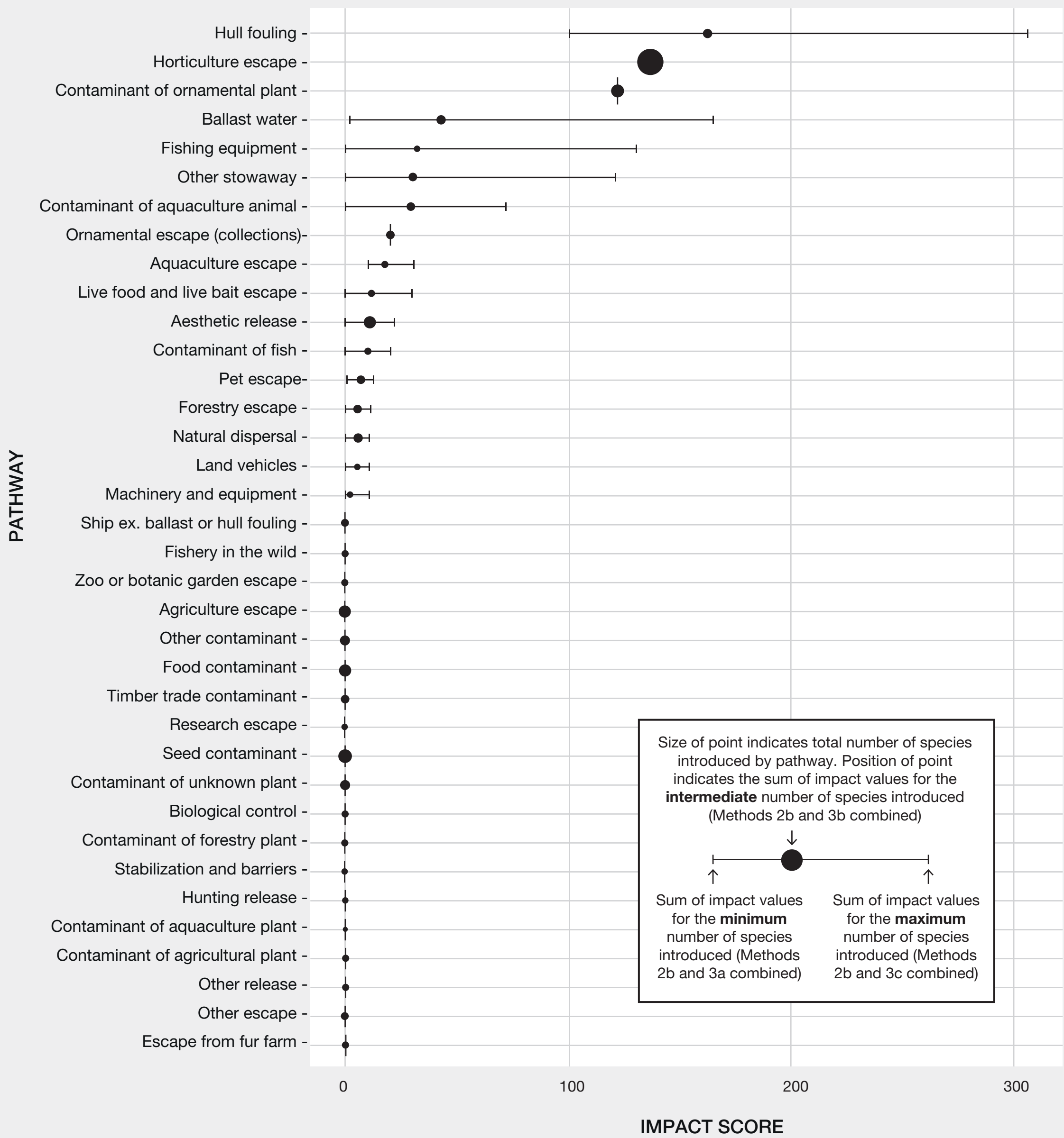


Figure 5 7 **Example of a pathway prioritization with a case study from Great Britain (United Kingdom).**

Pathway ranks using weighted impact scores for all established alien species in Great Britain. Point size indicates total number of species introduced since 1950, while position of points with error bars indicates the sum of impact values for the minimum, intermediate and maximum number of species introduced by each pathway since 1950. Source: Booy (2019), https://theses.ncl.ac.uk/jspui/handle/10443/4926, under license CC BY 4.0.

each pathway provided a pathway prioritization **(Figure 5.7)**. This was considered a more rigorous prioritization process than just using numbers of alien species per pathway, because pathway management is about reducing the risk of future arrivals and impacts.

b) Species prioritization

Caceres-Escobar *et al.* (2019) assessed the cost-effectiveness of six management scenarios for *Vulpes vulpes* (red fox) and *Felis catus* (cat) that were co-developed with local land managers and community groups on Minjerribah-North Stradbroke Island in Australia. Community prioritization of invasive alien plants was also undertaken in Chitwan-Annapurna Landscape of central Nepal using community memory of their arrival often due to a lack of knowledge of their impact status (Shrestha *et al.*, 2019).

Prioritization through horizon scanning is a prerequisite for deciding which species to consider for risk analyses. The European Union used horizon scanning to prioritize a list of alien species not yet present in Europe to inform selection of alien species for risk assessment and potentially future listing (H. E. Roy, Bacher, *et al.*, 2018). The list, published in 2018 partly (coupled with risk assessment) informed the list of invasive alien species of European Concern that underpins the Regulation on invasive alien species (European Union, 2014). Experts prioritized species based on likelihoods of i) arrival, ii) establishment, iii) spread and iv) magnitude of the potential negative impact on biodiversity and ecosystems over the next decade, within species thematic groups. From the 329 species initially considered, a final prioritized list was made of 66 species including eight species considered very high risk, forty species as high risk and 18 species as medium risk. A similar process was

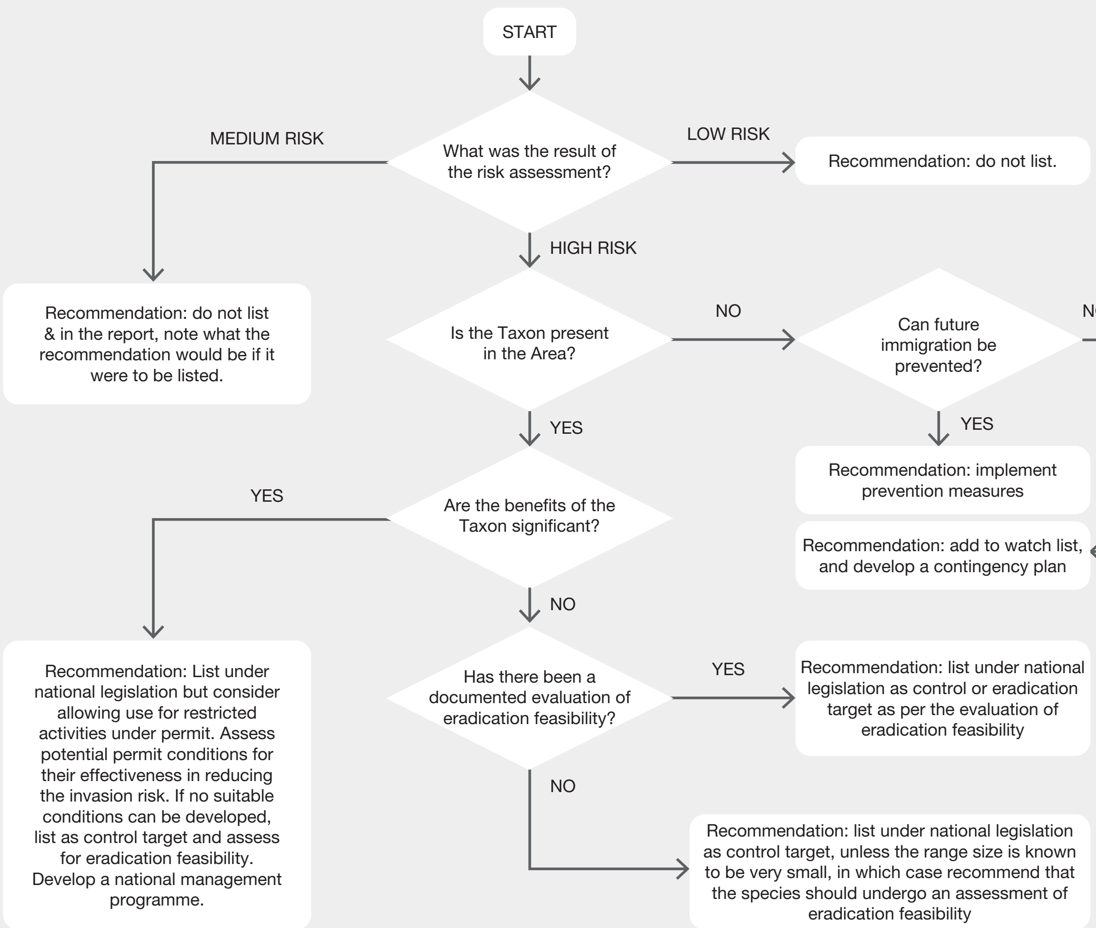


Figure 5 8 **Risk Analysis for Alien Taxa (RAAT) framework developed in South Africa as a standardized and transparent approach to prioritizing and regulating alien species based on evidence.**

The figure shows the process leading to the development of recommendations for the listing of alien taxa. Adapted from Kumschick, Foxcroft, *et al.* (2020), https://doi.org/10.1007/978-3-030-32394-3_20, under license CC BY 4.0.

undertaken in Australia where the task was to generate a National Priority List of Exotic Environmental Pests, Weeds and Diseases from which the top five to six species (from 168 initially identified) were classed as posing the greatest threat to the environment in each of eight biological groups including marine, freshwater and terrestrial ecosystems (ABARES, 2021).

Risk assessment: The Risk Analysis for Alien Taxa (RAAT) framework was developed in South Africa as a standardized and transparent approach to prioritizing and regulating alien species based on evidence (Kumschick, Foxcroft, *et al.*, 2020; Kumschick, Wilson, *et al.*, 2020; **Figure 5.8**). The aim was to increase capacity through expert and stakeholder workshops to prioritize species for regulation and management plan development. The framework has since been used retrospectively on species already regulated to confirm whether these should continue to be listed. Of 650 regulated species, 62 have been assessed, several of these now have a recommendation to change their regulatory status as they are not present in the country (delist) or can no longer be eradicated (move to widespread list). The regulators are now processing these recommendations *via* a committee and stakeholder consultation and are considering giving the framework legal force.

Risk management: On Viti Levu in Fiji, Daigneault and Brown (2013) undertook cost-benefit analyses of the management of five established species: *Spathodea campanulata* (African tulip tree), *Herpestes javanicus auropunctatus* (small Indian mongoose), *Papuana huebneri* (taro beetle), *Pycnonotus cafer* (red-vented bulbul) and *Decalobanthus peltatus* (Merremia). These analyses used survey data, impacts due to the species and management options. The cost-benefit analysis showed that benefits from management far outweigh the costs supporting the need to better manage invasive alien species in the Pacific, but that the most cost-effective management option varied between species.

c) Site prioritization

Prioritization of sites for invasive alien species management is built on the individual contexts of national environmental legislations (e.g., threatened species or ecosystem recovery and/or creating protected area networks), local knowledge, resources and management capacity and the cost-effectiveness of available management options. Managing invasive alien species in protected areas will also be prioritized based on the degree to which key ecosystems are invaded or at threat from invasion (Foxcroft, Pyšek, *et al.*, 2013; Giakoumi, Pey, *et al.*, 2019; X. Liu *et al.*, 2020). A combined site and species prioritization framework was developed and implemented in the Brazilian tropical and subtropical dry and humid forest in the Itatiaia National Park (Ziller *et al.*, 2020) assessing the level of biological invasion across four locations by 50 alien species. High priority was given to sites with high risk or in the early stages of biological invasion and low invasive alien species frequency. Krug *et al.* (2009) developed a prioritization scheme for the management of invasive alien plants in the Cape Floristic Region (South Africa). The identification of priority areas was based on weighted decision criteria, but the influence of the weighting on the outputs requires evaluation.

d) Management prioritization for species

Prioritization of management options for species rather than prioritizing species for management is also commonly undertaken for single or multiple species. A participatory decision-support software "Zonation" developed for this purpose on Reunion Island uses available spatial data on native species (**Glossary**) and invaded habitats to define conservation targets and provide projections at management-relevant scale, which helps to prioritize invasive alien plant management actions (Fenouillas *et al.*, 2020). Management priorities are defined based on three criteria: area accessibility; site history and likely intervention effectiveness.

Helmstedt *et al.* (2016) prioritized all eradication strategy options for invasive alien mammals across all Australian islands taking into account the complex decisions faced by managers. The optimal strategy was to eradicate a subset of invasive alien mammals, intentionally leaving some where either eradication costs were too high or removal might lead to complex ecological responses (e.g., trophic cascades). This eradication strategy was the most cost-effective generating 27 per cent greater ecological benefit across all islands compared to eradicating all invasive alien species on an island.

For marine invasive alien species where eradication is very unlikely, managing abundance to below ecologically defined impact thresholds is a better strategy (Usseglio *et al.*, 2017). Giakoumi *et al.* (2019) used experts to prioritize 11 management actions for 12 invasive alien species with different distributions and dispersal capacity. Each action was assessed using five criteria (effectiveness, feasibility, acceptability, impacts on native communities and cost) combined into an "applicability" metric. Rapid removal early in the biological invasion process and seeking commercial value from remaining species were ranked the highest management actions, while application of biological control ranked the lowest.

5.2.2.3 Dealing with uncertainty in decision-making

Decision-making, including for management of biological invasions, is weakened by multiple forms of uncertainty, bias and knowledge gaps (Moon *et al.*, 2017). Key gaps include future threats, likely-establishment patterns and the interactions with climate change (Leung *et al.*, 2012). Regan *et al.* (2002) developed a typology of uncertainty in conservation decision-making taking these gaps into account. This typology includes epistemic uncertainty associated with the knowledge of the system, and the linguistic uncertainty associated with communication between culturally different stakeholders. It has been used to evaluate the degree of uncertainty associated with prioritization approaches by McGeoch *et al.* (2012) **(Table 5.2)**. The majority of uncertainty sources are epistemic, and are caused by quantitative inaccuracies and knowledge gaps.

Decision-support tools generally explicitly assess the uncertainties around assumptions or knowledge (González-Moreno *et al.*, 2019; Leung *et al.*, 2012; Probert *et al.*, 2020), but not always (Caton *et al.*, 2018). Understanding bias and documenting and explaining uncertainty to decision makers and other stakeholders are critical in risk communication (Lundgren & McMakin, 2018; Probert *et al.*, 2022; WHO, 2013) and management decision-making (D. A. Clarke *et al.*, 2021; S. Liu *et al.*, 2011; Vanderhoeven *et al.*, 2017; A. I. Ward *et al.*, 2020). This allows a degree of confidence to be associated with decisions, increasing their legitimacy and providing transparency for managers (Estévez *et al.*, 2015; van der Bles, 2019).

5.2.2.4 Quantitative decision- support tools for implementing management options

Many quantitative decision-support models and platforms have been developed to support management implementation. Some generic modelling platforms have already been discussed (e.g., Tools for assessing pest and pathogen aerial spread (TAPPAS); Durr *et al.*, 2017; **section 5.2.2.1**). Their utility is broad and, when validated, can be cost-effective (M. E. Wilson & Coulson, 2016). Such platforms can be tailored to different manager perceptions or risk, types of invasive alien species and policy options (Lodge *et al.*, 2016; Perrings, 2016) and other management types such as pathway management (Leung *et al.*, 2014).

Such modelling platforms can help answering a wide range of risk-based management-related questions important to all stakeholder communities involved in a response to control an invasive alien species. Dynamic modelling platforms can be deployed during a management response to influence real-time decision-making. These tools have been applied, for example, for the eradication of foot-and-mouth disease (Garner & Beckett, 2005), pandemic influenza (Beckett, 2008), *Vulpes vulpes* (red fox), *Sus scrofa* (feral pig), *Felis catus* (cat; Ramsey *et al.*, 2011), *Trachemys scripta*

Table 5.2 **Dealing with uncertainty in decision-making for management of biological invasions.**

Types of knowledge (epistemic) and linguistic uncertainty, and errors associated with invasive alien species listing during the decision-making process that can be considered and documented when relevant. Adapted from McGeoch *et al.* (2012).

Type of uncertainty	Errors associated with alien species listing	
Epistemic uncertainty	Measurement error	Human error
		Incomplete information searches
	Systematic error	Species identification incorrect as a result of taxonomic uncertainty
		Survey information on presence, extent and population dynamics
		Resolution and scaling of invasive alien species range
		Data and knowledge not documented
		Documented data and knowledge not readily or widely accessible
	Stochasticity and natural variation	Survey information on presence, extent and population dynamics
	Subjective judgement	Baseline information on indigenous range
		Species designation as invasive
	Model uncertainty	Adequacy of research on impacts on biodiversity
Linguistic uncertainty	Vagueness	Species designation as invasive
	Context dependence	Resolution and scaling of alien range

elegans (red-eared sliders) (García-Díaz, Ramsey, *et al.*, 2017) and various weedy plant species (Panetta, 2012; T. J. Regan *et al.*, 2006; J. R. U. Wilson *et al.*, 2016). Publicly available tools are also under development for cost-effective decisions when eradicating invasive alien species (Centre for Invasive Species Solutions, 2021). Eradication programmes can have rule-of-thumb-based models or be dynamically assessed for likelihood of success (Panetta *et al.*, 2011; Panetta & Cacho, 2014). These help to ensure eradication programmes are neither terminated too early (Rout *et al.*, 2014) nor run beyond any real strong likelihood of success. Decision-making in the context of eradication programmes can also be assisted by Bayesian statistical methods (J. M. Keith & Spring, 2013; Solow *et al.*, 2008). Other methods include scenario tree analysis (Dominiak *et al.*, 2011) and Epitools (Sergeant, ESG, 2018). Predator-Free New Zealand has recently generated a rapid eradication assessment tool for invasive alien mammals (J. H. K. Kim *et al.*, 2020).

Pest risk maps are commonly used for strategic and tactical decision-support in managing biological invasions. However, such maps rarely measure spatial risk and are generally only used to estimate risk in one component of the invasion curve (general introduction or establishment risks – **Figure 5.1**), and can be improved to understand risks and consequences across the invasion steps and interdependencies (Camac *et al.*, 2020). More complex population-based modelling platforms can combine ecological distribution and climate data with process-based models to model pest establishment and spread, density and include impact risk analysis (e.g., Kriticos *et al.*, 2017; Z. Li *et al.*, 2016). These types of models can be made scalable from region down to farm level and provide risk-maps in near-real-time. Similar modelling tools also support decision-making around long-term management of invasive alien species and evaluating control programmes (Bourdôt *et al.*, 2018; Shephard *et al.*, 2016) as well as supporting ecosystem restoration to build resilience to prevent reinvasion. Most modern tools will have a mapping capability, and most will also use spatial information as a component of their evaluation (Beckett & Garner, 2007). Such tools can include individual-based (or agent-based) simulation models (e.g., Beckett, 2008), stochastic and deterministic mathematical models (e.g., Buckley *et al.*, 2005; Tildesley *et al.*, 2012) or, a combination of individual-based and mathematical approaches (e.g., Bradhurst *et al.*, 2015). High power computing helps draw inferences on invasive alien population change in space and time. Other model types include bioeconomic modelling, option value models, endogenous risk theory models, and other economic models. Many of these types of tools can also benefit from artificial intelligence to assist optimizing dynamic response approaches. Collectively, there are no fixed impediments to any of these forms of modelling, other than the availability of the relevant data, including spatial data and the time and investment required to design, implement and validate a model (**Chapter 1**, **section 1.6.7.3**).

5.3 TARGETING PATHWAYS, SPECIES AND SITES IN PRACTICE

5.3.1 When to implement pathway, species-based and site-based management strategies

As discussed in **section 5.1**, there are three main approaches for the management of invasive alien species: management of the pathways of biological invasion, management of the invasive alien species itself and site-based or ecosystem-based management. Pathway management approaches use methods to prevent incursions at the point of entry/border and post-border dispersal within jurisdictions (**sections 5.2.2.2**, **5.4**). Eradication, containment, or suppression of invasive alien species (control) are the main means of species-based management. The likelihood of successful species-based management usually declines with increasing distribution and density of the target invasive alien species **(Figure 5.1)**, except for classical biological control. Where the ability of a species-based programme to eradicate, contain or control the target invasive alien species is limited or where the emphasis may be on maintaining natural assets (e.g., threatened and endangered species or ecosystems), or on the maintenance of a site-based approach may be most likely to achieve long-term conservation outcomes, especially in terrestrial and closed water systems. This is particularly relevant for sites of high biodiversity and ecosystem significance in the context of nature's contributions to people and good quality of life conservation. Site-based approaches also aim to manage sites at risk from, or impacted by, multiple invasive alien species.

Site-based approaches are focused on delineated areas based on the values, objectives and environmental assets of the site. These delineated areas may include islands, protected areas, Indigenous sacred sites or other designated areas that contribute to good quality of life. Following site identification and prioritization, site-based management strategies generally include invasive alien species removal combined with site restoration in terrestrial ecosystems. At the ecosystem level, this is often described as ecosystem-based management. All three types of management approaches play key roles in management of biological invasions and are not mutually exclusive, therefore strategies and decision-making frameworks are needed to determine the context of when each management approach is best applied (Downey & Sheppard, 2006). Selection of the most appropriate approach depends on the outcomes sought and the available resources.

5.3.1.1 Implementing pathway management strategies

Pathway management can be applied to international pathways, which facilitate long-distance global invasive alien species dispersal (e.g., postal mail, trade, human travel, transport vessels, inland and marine canals; Hulme, 2009) and post-border domestic pathways (e.g., spread *via* agriculture or domestic trade, local travel and transport). Commodity-related drivers such as manufacturing, agricultural and pet trade shipping routes (including e-commerce; **Glossary**) and human-travel networks such as tourism and airline travel create invasion pathways (**Chapter 3**, **section 3.2.3.4**). Key components of pathway management include phytosanitary treatment of imported commodities and a combination of both active and general surveillance methods for early detection of invasive alien species to enable management outcomes to be achievable **(Figure 5.1)**.

Comparisons of general patterns of species introductions globally indicate that the commercial animal-trade (livestock, aquaculture introductions, companion animals and illegal pet trade), plant-trade (agricultural and horticultural commodities and trade in wood, seeds and ornamental and nursery stock), wood packaging and hitchhiker or contaminating pests and diseases arriving on other freight are the most significant pathways for terrestrial and freshwater species, whereas ballast water and hull biofouling are important invasion pathways for marine species (Downey & Sheppard, 2006; Hulme, 2009). International cooperation helps to understand pathway risks and manage long-distance pathways, through legislation, regulation, international guidelines and agreements (e.g., IPPC, WOAH), risk analysis, risk mapping, control of invasive alien species and mitigation of impacts (CBD, 2014; Hulme, 2009; Paini *et al.*, 2016). The IPPC has defined trade pathways of invasive alien species movement and provides standards on most plant trade pathways with respect to alien species movements through a range of ISPM for example the adoption of ISPM-15 in 2002 to manage wood boring insects in wood packaging material such as pallets has seen a reduction in incidence of invasive wood borers (Haack *et al.*, 2014). The exceptions are pathways of "contaminating pests" (i.e., "hitchhikers"), which spread through trade *via* movement of sea and air containers but are not associated with any specific commodities (IPPC-CPM, 2020) and e-commerce (Stringham *et al.*, 2021), but these are being

Table 5 3 **Management challenges and information needed when addressing invasive alien species risk associated with e-commerce.**

E-commerce is a rapidly increasing means of invasive alien species spread. Adapted from: CBD (2022c).

Management challenges	Information needed and implementation options
Risk	Improving information on the risks posed by e-commerce (including illegal e-commerce). Establish an international invasive alien species risk-based labelling system for shipments potentially containing invasive alien species as environmentally hazardous living organisms.
Commodities and invasive alien species	Identify commodities related to soils and growing media and living organisms. Create lists that specify which alien species may possibly be imported, including plants (and plant related), aquatic organisms, pet-trade.
Tools	Use autonomous internet tools to identify and locate e-commerce traders and other stakeholders. Gather data to monitor compliance and to evaluate the efficacy of risk mitigation measures. Apply non-intrusive inspection technologies and disseminate good practices and risk-based interventions using data analytics. Improve tools to support efficient international collaboration to link existing security initiatives with invasive alien species risk management and targeted (risk-based) inspections (databases and advanced digital supply chain management systems).
Communication and training	Better inform and communicate with all stakeholders and Indigenous Peoples and local communities in the early detection of incursion or spread of e-commerce derived invasive alien species in natural and managed ecosystems across traditional lands and waters. Develop voluntary codes of practices and standards to regulate cross-border e-commerce. Develop and implement training programmes and tools to facilitate appropriate levels of monitoring and inspection in e-commerce markets.
Management	Develop and apply improved management measures to minimize the risks of introduction of invasive alien species through e-commerce, consistent with international obligations.
Hazard identification	A substantial challenge is posed by living organisms currently being traded through e-commerce and whose risks have not yet been assessed.

addressed. While e-commerce is a key driver and pathway of international concern due to the increasing global volumes of parcel mail (**Chapter 3**, **section 3.2.3.1**), international efforts are underway to address this pathway (CBD, 2020b). **Table 5.3** illustrates options for implementing a coordinated e-commerce management programme.

Six categories of pathways of invasion have been recognized: release, escape, transport – containment, transport – stowaway, corridors and unaided (**Chapter 1**, **section 1.4.1**; **Chapter 2**, **section 2.1.2**, **Table 2.1**; **Chapter 3**, **section 3.1.1**). Deliberate releases, escape from confinement, containment of propagules (e.g., sanitary and phytosanitary control), prevention of stowaways and early detection and rapid response to combat natural spread from neighbouring regions need effective regulations at the jurisdictional level. Where jurisdictions do not have pathway management protocols in place invasive alien species will continue to establish (**section 5.5**), but where they are used effectively excellent pathway management can be achieved (e.g., **Box 5.2**). Various codes of conduct have been endorsed by the European Union for the

Box 5 2 **Case study: A successful pathway management programme from Barrow Island, Australia.**

One of the most ambitious and successful programmes of pathway management was the zero-tolerance biosecurity programme applied to protecting a class A nature reserve – Barrow Island, Australia (**Figure 5.9**; Merwe, 2015; Moro *et al.*, 2018; Scott *et al.*, 2017). The construction of a large liquefied natural gas plant required the transfer of material and personnel through marine vessels and aircraft to the island (**Chapter 3**, **section 3.3.2**). Since higher traffic brings a higher risk of introductions of invasive alien species (**Chapter 3**, **section 3.2.3.1**; **Chapter 2**, **Box 2.5**), a condition for the construction and operation of this project was that no alien species establish in the reserve. To date the Chevron pathway management programme for Barrow Island has been a success. The success of this biosecurity programme resulted from a risk analysis of all material and passenger pathways; identifying decontamination points and marine loading facilities; and all cargo undergoing pre-border cleaning, treatment, packaging and inspection of all transports and cargo (including a purpose-built low biosecurity risk container design) prior to transportation to the island. A quarantine system including behavioural incentives such as performance credits for all island personnel was established to prevent the establishment of terrestrial and marine alien species to the island and surrounding marine habitats. Marine invasions were the most challenging (e.g., Dias *et al.*, 2021). Seventy five percent of invasive alien species were detected pre-border (the majority on transport equipment or materials) which were invertebrates or seeds and 61 per cent detected post-border were *via* human assisted pathways on personnel and in luggage.

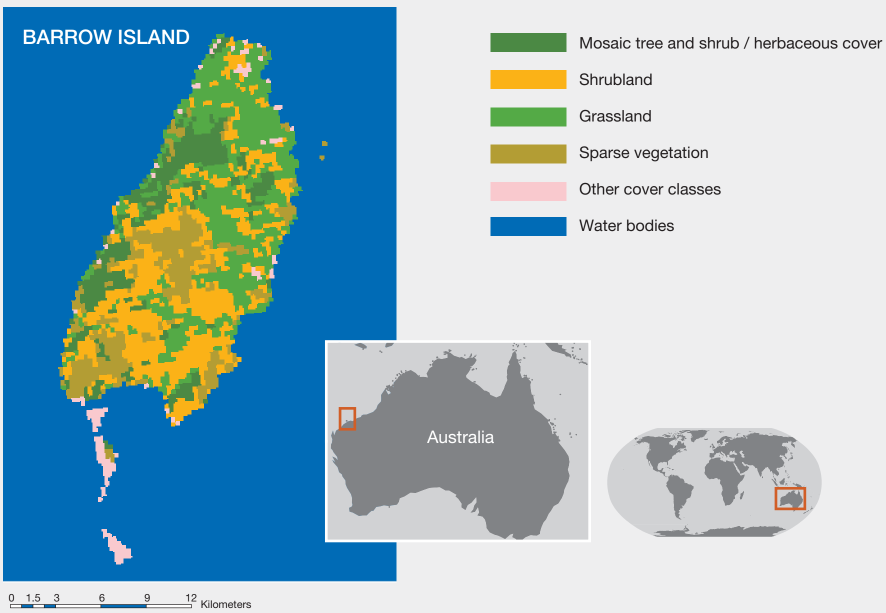


Figure 5 9 **Map of Barrow Island in north-western Australia which is dominated by grasslands and has an area of 202 km².**

pathway management of invasive alien species (Council of Europe, 2021).

Managing intentional introduction pathways helps preventing alien species that have been profiled through import risk analysis and that have high potential ecological impacts, reducing future unintentional spread and ecosystem impact risks (Pergl *et al.*, 2017). Understanding pathway risks through analyses of levels of trade between ecologically compatible countries and numbers of high-risk invasive alien species they do not yet share can help to better target pathway management. Risk can be quantified based on likelihood of arrival and establishment of whole complexes of invasive alien species (Banks *et al.*, 2015). An analysis of almost 1,300 known invasive alien insect pests and pathogens, based on total potential cost of these species invading each of 124 countries showed apparently climatically similar countries varying markedly in risk profile, depending on specifics of agricultural commodities and trade patterns (Paini *et al.*, 2016). According to the same study, the biggest agricultural producers were the greatest potential sources of invasive alien species but could also experience the greatest cost from future biological invasions. Similarly, data from border interceptions, trade volumes, country pest occurrence records and climate suitability models can be used to develop models to estimate the exposure risk of potential and current trading partners leading to an established population of a new high threat pest or disease (Camac *et al.*, 2021). A pathway-centred conceptual model has also been used to determine the role of pathways in invasive alien species establishment and design early detection and rapid response programmes (Colunga-Garcia *et al.*, 2013).

In several regions, control actions have reduced numbers of species deliberately released and to some extent escapes, although species continue to be introduced unintentionally as contaminants and stowaways (Hulme *et al.*, 2008). For example, more than 400 metazoan introductions were reported to have spread through the Suez Canal (Galil *et al.*, 2021), and of 1,257 alien marine species in Europe, shipping (Katsanevakis *et al.*, 2013) and the Suez Canal (Galil *et al.*, 2021) were likely responsible for increasing introductions. The freshwater fish *Pseudorasbora parva* (topmouth gudgeon), spread across Europe, was a contaminant of commercially-exported fish consignments (Gozlan *et al.*, 2010; **Chapter 3**, **section 3.2.3.2**).

5.3.1.2 Implementing species-based surveillance and management

Surveillance aims to detect new invasive alien species incursions early enough to allow for an effective rapid response towards eradication (**section 5.4.2**). Active surveillance is designed to detect priority invasive alien species to inform pathway risk assessment and to provide prevalence information on a trade pathway or a delimited area containing a suspected incursion (IPCC, 2018; **Supplementary material 5.8** for more details on surveillance guidelines). Terrestrial, aquatic and animal disease surveillance (IPPC, 2018; World Organisation for Animal Health, 2019) is generally focussed on specific threats and aims to demonstrate absence (i.e., supporting trade) or to detect prevalence at low levels to ensure rapid response and eliminate the disease or pest outbreak. Surveillance programmes are underpinned by a well-developed sampling methodology and statistical design, which provide transparency around confidence and detection thresholds (Kalaris *et al.*, 2014; FAO, 2018a; World Organisation for Animal Health, 2019). Stochastic scenario tree models can be used to describe each component of the surveillance system to demonstrate that a zone or country is free from a particular disease (P. A. J. Martin *et al.*, 2007). Online calculators such as Epitools assist the design of animal surveillance programmes demonstrating disease freedom (P. A. J. Martin, 2008; Sergeant, ESG, 2018). Stochastic scenario tree modelling of each of the surveillance system components can be used to estimate the probability of disease freedom (Sergeant, ESG, 2018) and to test the sensitivity of the surveillance system. For example, scenario tree modelling was used to assess the sensitivity of Ecuador's national surveillance system to human leptospirosis by conducting probabilistic modelling for each component of the surveillance system. The model assessed the programme's sensitivity as an output so that an economic assessment of the system could be made (Calero & Monti, 2022). Another example of use of stochastic scenario tree modelling is in helping planning a surveillance programme to demonstrate disease freedom for *Mycoplasma bovis* in cattle after an extensive and costly eradication programme in New Zealand (Cowled *et al.*, 2022).

Integrated evaluation frameworks and tools also help evaluate surveillance systems (Peyre *et al.*, 2019). The Food and Agriculture Organization (FAO) Surveillance Evaluation tool is part of the Emergency Prevention System for Animal Health providing countries with comprehensive and standardized methods to evaluate animal disease surveillance including zoonoses and action plans to track diseases that affect animals and people (Aguanno *et al.*, 2019).

Following detection of a new priority invasive alien species, rapid response can only be achieved with immediate access to resources, as the time it takes to mount an effective response is generally of limited duration. In most jurisdictions there is a lack of legislation, policy, protocols or plans to guide rapid management responses to new incursions. Many countries are still establishing such systems, but these are currently seldom implemented or require support from donor agencies (Boy & Witt, 2013). There are good working policies in some countries where pre-negotiated rapid-response plans are agreed at a species-level before each incursion is detected (**section 5.2.2.3**). Such plans pre-negotiate roles and identify funding and responsibilities around species prioritized as key future threats. Where the chance of invasive alien species eradication is lost, management can be done through site-based management, but it is more costly (e.g., *Sciurus carolinensis* (grey squirrel) in Europe; Bertolino & Genovesi, 2003).

Large scale species-based removal and eradication programmes have produced successful results, for example, on islands and for mammalian invasive alien species in northern Europe (Robertson *et al.*, 2017). Species-based management is more likely to achieve impact if relevant stakeholders collectively agree and clearly define overarching management objectives beyond species suppression (a reduction in the abundance of an invasive alien species population). These overarching management objectives could include objectives to measure benefits in biodiversity and ecosystem services or the reduction of threats to threatened and endangered species and communities. Managing invasive alien species in marine environments is particularly challenging, and some species-based management approaches were attempted on sun corals but only reduced localized colonies at small scales in the short term **(Box 5.3)**. Managing freshwater biological invasions is also challenging. In Indonesia, the main invasive alien freshwater fish include species of *Pygocentrus nattereri* (red piranha), *Tetraodontidae* spp. (pufferfish), *Trichomycteridae* spp. (parasitic catfish) and *Electrophorus electricus* (electric eel; Francis, 2011). They were brought in as part of a very large ornamental fish farming sector, and then escaped in rivers on many of the Indonesian islands, causing significant impacts on native freshwater communities. Indonesia has now taken a species-based approach which actively regulates the movement of these alien species within the Indonesian archipelago, and has banned 30 alien fish species from importation (Priono & Satyani, 2010). In Arizona, United States, successful invasive alien fish management has been achieved (e.g., *Salmo trutta* (brown trout)) through long-term collaboration between government agencies and the Indigenous White Mountain Apache tribe using cultural beliefs and habitat restoration practices leading to increases in the native *Oncorhynchus apache* (apache trout) populations (Pfeiffer & Voeks, 2008). In the People's Republic of China, while a biological control programme is under development, national containment lines with 30 km buffer zones were proposed for *Ageratina adenophora* (Croftonweed), to prevent spread from Yunnan province in the south west to other provinces to the north and east (Wan *et al.*, 2009).

Some widely established invasive alien species **(Figure 5.1)** can be targeted using classical biological control (**sections 5.4**, **5.5**) aimed at suppressing populations (number of individuals) at local and landscape levels. There have been over one hundred successful programmes using biological control against invasive alien plants (Schwarzländer *et al.*, 2018). For example, a survey with local communities in Eastern Africa showed *Opuntia stricta* (erect prickly pear) contributed to the loss of grazing land and health impacts (e.g., mouth sores, weight loss and death) of livestock but only 20 per cent of respondents could attempt manual control (R. T. Shackleton *et al.*, 2017). The subsequent release of the *Opuntia stricta* specific genotype of *Dactylopius Opuntiae* (prickly pear cochineal) as a biocontrol agent led to very effective management. In Tahiti, biological control using the fungus *Colletotrichum gloeosporioides* f. sp. *miconiae* of the pan-pacific invasive alien plant *Miconia calvescens* (miconia) from South America has effectively broken the complete canopy cover of *Miconia calvescens* allowing native species to return, but manual removal is still important in ongoing ecosystem restoration (Meyer, 2008).

A review of 76 relevant case studies suggested that the majority of the management conducted by Indigenous Peoples and local communities is species-based (**Supplementary material 5.1**). Therefore, some Indigenous Peoples and local communities have developed knowledge and culture that are critical for motivating species-based actions and prioritizing targets, in many cases utilizing available resources as part of local management. In Canada, *Fraxinus nigra* (black ash) is threatened by the invasive alien beetle *Agrilus planipennis* (emerald ash borer). The Indigenous Kahnawake People use *Fraxinus nigra* trees for basket making, which has increased the public demand for conserving *Fraxinus nigra* (IPBES, 2020). In Hawaii, traditional gatherers of native ferns for cultural practices incorporate manual control of invasive alien plants to manage the fern resource (Ticktin *et al.*, 2006). In a different approach, management can be done through utilization of targeted invasive alien species. For example, the Indigenous Maya Kaqchikel community in Guatemala has recognized the negative impacts of *Pseudopanax laetevirens* (sauco tree or saúco cimarrón in Spanish) and community control efforts have included developing alternative uses for *Pseudopanax laetevirens*, including in food and medicine, which has improved awareness of the benefits and impacts of the tree, helping to limit its spread (IPBES, 2020). Similarly, the loss of native vegetation for livestock feed in various local communities in East Africa (Kenya and Tanzania) from the invasion of the *Prosopis juliflora* (mesquite) tree since the 1970s led to the development of alternative uses of it for firewood and livestock food supporting livelihoods (**Chapter 4**, **Box 4.9**). Nonetheless, spread has continued unabated (Mbaabu *et al.*, 2019) and *Prosopis juliflora* has been declared as a major invasive alien species in Ethiopia, Kenya, India, South Africa and the Sudan (Chandrasekaran & Swamy, 2016; R. T. Shackleton *et al.*, 2014).

Box 5 3 **Case study: Species-based management of invasive alien corals through resource use in Brazil.**

Tubastraea spp. (sun corals; **Figure 5.10**) are highly invasive and widely spread along the Brazilian coast, where, at some locations, they occupy 80 per cent of the shallow subtidal seabed (Mantelatto *et al.*, 2020). *Tubastraea tagusensis* forms dense clusters with up to 872 colonies per m^2 (Paula & Creed, 2005; de Oliveira Soares *et al.*, 2018), and has been recorded from depths of up to 40m (Figueroa *et al.*, 2019). *Tubastraea micranthus* (black sun coral) and *Tubastraea coccinea* (orange-cup coral) has been recorded at 138m and 90–96m below sea level, respectively (Sammarco *et al.*, 2013). *Tubastraea* spp. are considered to have spread with shipping and offshore oil infrastructure. Mantelatto *et al.* (2020) recorded the occurrence of *Tubastraea coccinea* and *Tubastraea tagusensis* attached to floating wood debris and marine litter indicating rafting over long distances may be another mechanism of range expansion. Genetic analysis of these species revealed multiple invasions, secondary introductions, and clonality (Capel *et al.*, 2019). The species-based management goal has been to slow the spread and reduce the negative impacts (Creed *et al.*, 2017). More than 231,000 sun coral colonies (about 8.3 tonnes along the coast of Rio de Janeiro) have been manually collected by trained divers using standard protocols. While preventing dispersal across extensive areas or coastlines was not feasible, focused removal and harvesting efforts provided value by generating income for coral harvesters (Creed *et al.*, 2017). Creed *et al.* (2021) documented manual removal as a recommended option to control and slow the spread and/or eradicate *Tubastraea* spp., however, *Tubastraea* spp. are widely spread in western Atlantic (Gulf of Mexico, Caribbean, Brazil), occur in dense clusters, and extend to depths beyond accessibility through recreational diving and are also nearly year-round prolific reproducers. Dispersal vectors are ubiquitous, which may assist colonization from surrounding areas. Although used as a resource, containment or controlling these species is unfeasible, even at local scale. This really questions the tractability of manual removal-based eradication of these species (Sammarco *et al.*, 2013).

Figure 5 10 **Invasive alien coral *Tubastraea* spp. (sun coral or coral-sol in Portuguese) off the Brazilian coast.**

The colony on the right has been manually removed as part of a species-based management programme. Photo credit: Joel C. Creed, Projeto Coral-Sol/UERJ – under license CC BY 4.0.

Alternative uses of invasive alien species resources have also been adopted in freshwater ecosystems (e.g., invasive paiche *Arapaima gigas* (arapaima) in the Bolivian Amazon; Macnaughton *et al.*, 2015) and marine ecosystems (e.g., *Paralithodes camtschaticus* (red king crab) in Finnmark; Broderstad & Eythórsson, 2014). Some local communities derive local names for some invasive alien species based on their impacts, which can assist recognition and understanding of the different invasive alien species in their areas (IPBES, 2020). Similarly, Indigenous herders in central Uganda use local names for plants in the area, including invasive alien species, which helps monitoring biodiversity (Oba *et al.*, 2008). In southern Tanzania, an invasive alien plant education and awareness campaign improved local appreciation of undesirable impacts, and voluntary manual removal together with basic equipment and the provision of seedlings of alternative desirable plant species provided community benefits (Foxcroft, Witt, *et al.*, 2013). Indigenous

Peoples and local communities can also assist in mitigation measures such as native seed collection, storage and restoration. In southern India, craftsmen harvest *Lantana camara* (lantana) for furniture and basket making which reduces local *Lantana camara* density and size classes (Kannan *et al.*, 2016), but beyond the villages, large regional scale abundance cannot be managed by harvesting.

5.3.1.3 Implementing site-based and ecosystem-based management programmes

Site-based management is likely to include removal of invasive alien species present in a site to achieve ecosystem restoration objectives in terrestrial and inland aquatic ecosystems. Site-based management is sometimes termed "asset protection" since it generally includes site revegetation and restoration (either towards the original or some new desired state) to increase site value and resilience to future invasion (Downey & Sheppard, 2006). Site-based management has been categorized into "susceptible" and "sensitive" sites (McGeoch, Genovesi, *et al.*, 2016). "Susceptible sites are those "with the greatest exposure to invasive alien species propagules and a high probability that these propagules will establish in the area", whereas sensitive sites are those "exposed to the greatest invasive alien species impacts" (McGeoch, Genovesi, *et al.*, 2016). To evaluate progress towards site-based management objectives of reducing community and ecosystem level impacts, ongoing monitoring is critical.

Site-based management is primarily focussed on a particular geographic location, while ecosystem-based management is focussed on a higher level of particular impacted ecosystems. For example, ecosystem-based management could include managing river flow regimes at the catchment scale to keep a myriad water bodies and riparian wetlands healthy and dominated by native species. Such hydrological management can be local (e.g., watering directly) or regional (managing environmental flow allocation) thus affecting multiple sites or ecosystems (Catford *et al.*, 2011, 2014; Ruhi *et al.*, 2019). Both site- and ecosystem-based approaches are on the same continuum defined by the objective(s) of the management, and the location and type of management actions needed to achieve those objectives. Similarly, management of whole socioecological systems at larger scales is also undertaken **(Box 5.2)**. In some contexts (e.g., United Nations Educational, Scientific and Cultural Organization (UNESCO) Man and Biosphere reserves; **section 5.3.2**), site-based management is approached from a socioecological systems perspective (**Chapters 1** and **6**). One example is the management of invasive alien plant (Jellinek *et al.*, 2014) and fishery resources in the Galápagos Islands marine reserve (Castrejón *et al.*, 2014; **Box 5.5** in **section 5.3.1.4**). For areas with limited biodiversity information, site-based management objectives may be expressed in terms of habitat, which facilitates the understanding, conservation or restoration status and economic value of the ecosystem (Dymond *et al.*, 2008).

Sites prioritized for management by Indigenous Peoples and local communities are likely to be sites where there already is an integrated management of culturally important sites and values with conservation outcomes or active community involvement in the control of invasive alien species (Bach *et al.*, 2019; **Chapter 2**, **Box 2.6**). Indigenous lands cover more than a quarter of the world's terrestrial area (Garnett *et al.*, 2018) and the relationship between invasive alien species, site cultural value and negative socioecological effects are often highly complex, contextual and often contradictory (Howard, 2019; Pfeiffer & Voeks, 2008). Sites of high cultural value may be valued for provision of food and medicine because they are also biodiversity refuges for native species. For example, Aboriginal-owned freshwater billabongs in northern Australia have cultural assets as hunting and fishing grounds, and therefore site-based management to exclude feral animals is being experimented collaboratively by researchers and Aboriginal people (E. Ens *et al.*, 2016). Invasive alien species have been incorporated into local cultural systems of Indigenous Peoples and local communities, leading to cultural enrichment (Pfeiffer & Voeks, 2008). The Lower Mekong Basin is a large-scale socioecological system where sites and invasive alien species are managed in an integrated manner (Miththapala, 2007). Indigenous Peoples and local communities utilize invasive alien species as a natural resource in the process of management **(Box 5.4)**.

5.3.1.4 Integrating pathway, species-based and site-based management

Pathway, species- and site-based management can be implemented at various spatial scales along the invasion continuum (**Figures 5.1** and **5.4**; **Box 5.4**). Integrating the use of pathway, species-based and site-based management can promote more informed resource allocation and decision-making (McGeoch, Genovesi, *et al.*, 2016). Integrated use of pathway, species-based and site-based management strategies can be implemented in larger, socioecological complex systems such as the Galápagos Islands **(Box 5.5)**. Differences in societal perceptions and values, impacts and management responses to invasive alien species can decrease the likelihood of success, a principle referred to as socioecological incompatibility (Beever *et al.*, 2019; **Chapter 1**, **section 1.5.2**; **Chapter 4**, **section 4.6**). Management therefore needs to include inter-agency, multi-stakeholder community cooperation from local to national levels for successful outcomes (van Wilgen *et al.*, 2020) if such integrated programmes are to be fruitful.

Box 5 4 Case study: Management of biological invasions in a socioecological system in Asia: the case of the Lower Mekong Basin.

The Mekong River flows through six countries (China, Myanmar, Thailand, Lao People's Democratic Republic, Cambodia and Vietnam), draining an area of 795, 000 km^2. The Lower Mekong Basin is a biodiversity hotspot with numerous endemic and endangered species, and home to about 60 million people, some of whom are Indigenous Peoples and local communities (Miththapala, 2007). Invasive alien species (e.g., *Pontederia crassipes* (water hyacinth), **Figure 5.11**) are impacting biodiversity (e.g., the invasive alien *Mimosa pigra* (giant sensitive plant) displacing native wetland species), human health (e.g., the invasive alien *Pomacea canaliculata* (golden apple snail) vectors and the nematode *Angiostrongylus cantonensis* (rat lungworm) causing eosinophilic meningoencephalitis in humans) and causing severe impacts on important food resources (e.g., rice). As part of the Mekong Wetlands Biodiversity Conservation and Sustainable Use programme (Friend, 2007), a multi-national biological invasion strategy was developed for the whole Lower Mekong Basin and has been implemented at national and local levels, focusing on 14 invasive alien plants and 15 animals (including 10 invasive alien fish and three invasive alien snails). The strategy includes a) pathway management by preventing further entry of invasive alien species and controlling the spread of priority invasive alien species, especially in protected areas; b) increasing public awareness and support (in local languages for communities dependent on the Mekong River); c) building capacity and strengthening national and regional policies and legislation; d) identifying alternative uses for invasive alien species to support control and providing additional benefits; e) evaluating economic impacts of invasive alien species; and f) developing early detection and rapid response and monitoring systems. However, the impacts from the construction of hydropower dams and increasing saline water intrusion have continued to cause degradation of the Mekong delta (Chua *et al.*, 2022; E. Park *et al.*, 2022; Sor *et al.*, 2020; Soukhaphon *et al.*, 2021).

Figure 5 11 *Pontederia crassipes* (water hyacinth) on the Mekong River.

Pontederia crassipes can impact on the livelihoods of Indigenous Peoples and local communities along the Mekong River. Photo credit: Pham Quang Thu and Colleague – under license CC BY 4.0.

Box 5 5 Case study: Integrated pathway, species-based and site-based management in the Galápagos Islands, Ecuador.

The Galápagos Islands is a World Heritage site due to its exceptional levels of endemism (Toral-Granda *et al.*, 2017). Although geographically isolated, at least 1,579 alien terrestrial and marine species have been introduced in the Islands, of which 1,476 have become established. From the arrival of the first people in 1535 until 1975, alien species arrival accelerated from an average of less than one species to about 30 new species per year, with half of them being intentionally introduced. These unintentionally and intentionally introduced species include 687 terrestrial plants, 17 animals for agriculture and 11 pet species. Unintentional plant contaminants (including seeds and plant-associated material) included 196 insects, 11 other terrestrial invertebrates, 53 marine invertebrates and 127 terrestrial plants. An integrated pathway (I. Keith *et al.*, 2016), species-based and site-based management plan could facilitate a comprehensive approach to the management of biological invasions to, and within, the Galápagos Islands. For example, pathway management could address external arrivals and movements between the islands (Veitch & Clout, 2002). Cargo quarantine and inspection is currently undertaken at a

Box 5.5

single facility supported by a centralized database of plane, boat, residents, tourists and cargo arrivals analysed to evaluate strengths and weaknesses in the control system. Efficient pathway management would include a marine biosecurity programme and developing regulations (Carlton *et al.*, 2019; Toral-Granda *et al.*, 2017). Educational programmes for residents and tourists, and risk assessments of human mobility and associated transports provide further information on how to manage pathways. Site-based management for plants is applied to terrestrial protected areas and urban centres to maintain remnant habitats and native biodiversity. A number of species-based programmes focus on species prioritization and priority plant (Gardener *et al.*, 2010) and mammal eradication (Cayot *et al.*, 2021) and biological control (Zachrisson & Barba, 2020; **Figure 5.12**).

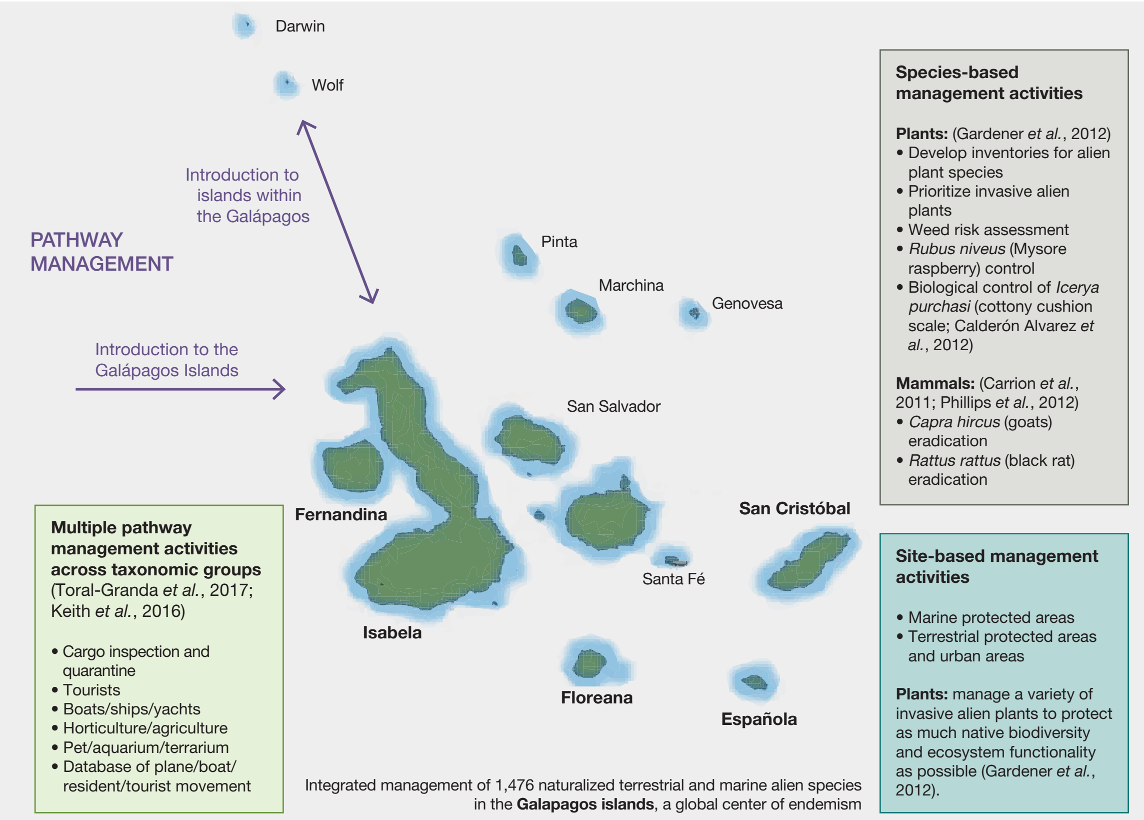


Figure 5.12 **Map of the Galápagos Islands showing examples of pathway, species-based and site-based management activities.**

These islands are a global centre of endemism where 1,476 naturalized terrestrial and marine alien species have been recorded (Calderón Alvarez *et al.*, 2012; V. Carrion *et al.*, 2011; Gardener *et al.*, 2012; I. Keith *et al.*, 2016; Phillips *et al.*, 2012; Toral-Granda *et al.*, 2017). Source of underlying map from: Andrew Z. Colvin, WM Commons – under license CC BY-SA 4.0.

5.3.2 Managing invasive alien species impacts in protected areas, islands, national parks, Ramsar Sites, Man and Biosphere reserves and World heritage sites

A Global Invasive Species Programme (GISP) report (De Poorter, 2007) covering largely terrestrial sites identified 487 protected areas with an invasive alien species threat, while a different study listed 135 protected areas which had a range of science-based management and monitoring programmes in place (Foxcroft, Pyšek, *et al.*, 2013). There are examples of successful species-based and site-based invasive alien species management programmes in protected areas. For example, a study that reviewed the status and outcomes of control over a 30-year period in 24 nature reserves across savanna, arid environments, islands and Mediterranean type ecosystems (Usher, 1988) showed that invasive alien mammals were decreased by 43 per cent after 30 years, but invasive alien plants continued to pose the greatest threat, increasing in 31 per cent of the nature reserves studied (R. T. Shackleton, Foxcroft, *et al.*, 2020). **Chapter 4**, **sections 4.3.1.2** and **4.4.1.2** discuss impacts in protected areas and **Chapter 6**, **section 6.3.1.4(5)**, discusses the need to incorporate management of biological invasions into protected area management plans.

National Parks are also often reservoirs of invasive alien vertebrates because of lack of resources to control them, which often raises concern for surrounding landowners. Mountain reserves have typically been considered resistant to invasions, however studies forecast an increase of invasions due to climate warming and anthropogenic related activities, including the expansion of tourism (Kueffer *et al.*, 2013). Pathway, species- and site-based management approaches can be applied by minimizing general access to wilderness areas, thereby reducing dispersal pathways **(Box 5.6)**. Globally, most protected areas are reliant on income from tourism (Meyerson & Reaser, 2002), a driver that promotes biological invasions (**Chapter 3**, **section 3.2.3.4**), to achieve their mandate and resource the management of biological invasions and reintroduction of native species (**section 5.5.3**). Pathway management may therefore have to account for vehicles and yachts (L. G. Anderson *et al.*, 2015), horses (Pickering & Mount, 2010), trail running (K. Smith & Kraaij, 2020) and tourist associated infrastructure such as in Masai-Mara National Reserve, Kenya (Witt *et al.*, 2017) and Kruger National Park, South Africa (Foxcroft *et al.*, 2019). Surveillance can be directed to areas of heightened concern, such as along roadsides (Pauchard & Alaback, 2004), in developed areas (including staff and tourist facilities) and disturbed areas. In the European Union, the Emerald Network of Areas of Special Conservation Interest formed under Natura 2000 aims to integrate these approaches for management of biological invasions across European designated protected areas (Bartula *et al.*, 2011; Kati *et al.*, 2015; **Chapter 6**, **Box 6.8**).

The Global Wetland Outlook (Ramsar Convention on Wetlands, 2018) indicated that in 2018, 40 per cent of the parties reported a comprehensive national inventory of invasive alien species impacting wetlands. However, few (26 per cent) had developed policies or guidelines to manage invasive alien species in wetlands (Ramsar Convention on Wetlands, 2018). The aim of management of biological invasions in Ramsar sites is to prevent water quality deterioration and facilitate use of the wetland substrate as resources for people in addition to biodiversity protection (Ramsar Convention Secretariat, 2010). Ramsar guidelines recommend prevention, eradication and control, by focusing on pathway, species-based and site-based management (Ramsar Convention Secretariat, 2010). Invasion by *Pontederia crassipes* (water hyacinth) in Malagarasi-Muyovozi (Tanzania) affects the livelihoods of local fisherman communities as fishing camps were closed (Kalumanga, 2015). Nyul Nyul rangers in the Kimberly region of Western Australia manage feral animals and invasive alien plants on their wetlands (The Commonwealth of Australia, 2016). In Beung Kiat Ngong Ramsar Site (Lao People's Democratic Republic) locals had to deal with the socio-economic impacts of *Pomacea canaliculata* (golden apple snail) by harvesting and selling them (Cranmer *et al.*, 2018).

A review of 241 World Heritage Sites identified 290 invasive alien species as a threat (R. T. Shackleton, Bertzky, *et al.*, 2020). For example, a management programme was recommended in 2006 for *Mimosa pigra* (giant sensitive plant), which is considered the largest threat to the biodiversity of Tonle Sap Biosphere Reserve (north-west Cambodia), a highly important floodplain habitat for fish and endangered waterbirds in South-East Asia (Goes, 2005). Widespread management at landscape scales was considered unfeasible (Ferguson & Chun, 2011), but as the species is river-dispersed, the programme recommended increased surveillance targeting *Mimosa pigra* and a basin-wide plan to reduce the risk of further introduction and establishment of other invasive alien species (van Zalinger, 2006). *Mimosa pigra* management in Kakadu National Park (World Heritage and Ramsar site) in Australia using classical biological and integrated control has provided long-term control. Effort is now turning to management of *Urochloa mutica* (para grass), *Hymenachne amplexicaulis* (hymenachne) and *Andropogon gayanus* (tambuki grass) (Setterfield *et al.*, 2013). Classical biological control of *Pontederia crassipes* (water hyacinth) is also underway in the Delta du Senegal (World Heritage and Biosphere reserve), Senegal (Amer *et al.*, 2015).

Islands are areas of special concern for management. On islands that are susceptible to invasive alien species introduced by trade and human movement (**Chapter 2**, **Box 2.5**; **Chapter 3**, **section 3.2.3**), a key strategy is to prevent the establishment of introduced invasive alien species. As human activities expand into more remote regions, including the Arctic, Antarctica and the South Atlantic and Pacific, biogeographic dispersal barriers are weakening (e.g., the Tristan da Cunha islands; D. Moser *et al.*, 2018) and rigorous biosecurity programmes are extremely important. The sub-Antarctic islands fall almost entirely in protected areas but have had numerous introductions of invasive alien species (Convey & Lebouvier, 2009; Frenot *et al.*, 2005). As a result, biosecurity measures have been generally implemented by the five sovereign nations to reduce future introductions of invasive alien species and undertake eradications and other management (Chown *et al.*, 2012; **Chapter 6**, **section 6.3.3.1**), which has led to increased awareness of biosecurity across all stakeholders. Elsewhere, various Small Island Developing States (SIDS) are also initiating successful biosecurity campaigns with good results (**Boxes 5.7** and **5.8**).

Invasive alien species are a major driver of species extinctions on islands (Sax *et al.*, 2002; Simberloff *et al.*,

Box 5 6 Case study: Management and use of *Sus scrofa* (feral pig) and *Axis axis* (Indian spotted deer) in El Palmar National Park, north-eastern Argentina, by local communities.

Both *Sus scrofa* **(Figure 5.13)** and *Axis axis* are considered to be a major threat within invaded ranges around the world. They impact on plant community structure and dynamics, compete with native grazers and livestock and may transmit zoonotic pathogens. In the El Palmar National Park in north-eastern Argentina created in 1965 to preserve one of the last high-density stands of the *Butia yatay* (yatai palm tree), *Sus scrofa* reduced recruitment by consuming fruits or seeds and killing saplings (Ballari *et al.*, 2015). The park rangers' initial efforts to cull *Sus scrofa* in 1983 were unstructured. A revised management programme based on hunting with trained dogs and spotlight hunting from the back of slow-moving vehicles initiated in 1995 was successful for *Axis axis*, but again proved unsustainable for the *Sus scrofa* population which continued to increase. Valuable lessons were learnt forming the foundation of a new multi-stakeholder management programme incorporating sustainability, broad social participation, safe procedures, close supervision and a regulated framework targeting both species in 2006. Controlled shooting teams worked uniformly across the park, without catch quotas. Each hunter was allowed to take home most of each carcass to minimize selective hunting and the rest were donated to local public schools, community shelters and retirement homes (Gürtler *et al.*, 2017).

This programme reduced *Sus scrofa* abundance within two years to levels causing minimal soil damage (Gürtler *et al.*, 2017). Recruitment rate of yatay palm trees significantly increased a decade later. *Axis axis* numbers however continued to increase, the reasons for which remain unclear (Gürtler *et al.*, 2018).

Figure 5 13 ***Sus scrofa*** **(feral pig, jabalí in Spanish) in El Palmar National Park where a management programme was implemented to control the invasive alien species.**

Photo credit: Alfredo Sabaliauskas (@sab.alfred) – under license CC BY 4.0.

Box 5 7 Case study: Biosecurity in the Republic of Seychelles.

Trade and travel increased the threats of biological invasions to the Seychelles archipelago although there were important weaknesses in the biosecurity policies for trade (Rocamora, 2015). Under an Environment Management Plan project, a new Biosecurity Service was created with strengthened technical and institutional capacities which helped the development of an emergency plan and operational manuals (Senterre & Dine, 2022). The entry and internal movement of animals and plant pests and diseases was regulated leading to improvements in the conservation status of native species. An unexpected result was that Seychelles was able to join the World Trade Organization (WTO) due to the strengthening of its biosecurity institutions, policy and legislation. Project challenges included finding qualified staff and consultants, creating a cost-recovery mechanism to support the Biosecurity Service and the creation of a group to coordinate knowledge management and information sharing at the national level (GEF, 2007).

2013; **Chapter 4**, **section 4.3.1**), however, eradication and control of invasive alien species on some islands, especially vertebrates, has been highly effective with rapid biodiversity benefits (Howald *et al.*, 2007; H. P. Jones *et al.*, 2016; Genovesi, 2011; **section 5.5**). On the Motuopao Island (New Zealand), a species-based control programme of invasive alien plant species assisted native grasslands to recover following the control of *Malva arborea* (tree mallow; Beauchamp & Ward, 2011). Holmes *et al.* (2019) identified 169 globally important islands where invasive alien mammal eradications would assist threatened vertebrate species. This was based on a conceptual framework considering biogeographic (i.e., extinction risk, irreplaceability, severity of impact from invasive alien species) and technical feasibility of eradication (i.e., operational cost of the programme, size of the island, no permanent human settlements) as well as socio-political feasibility to initiate an invasive mammal eradication project by 2020 or 2030. The list included some SIDS such as Bermuda, Cape Verde, Cuba, Fiji, Kiribati, Palau and the Seychelles.

Island eradication programmes employing species-based approaches focus on the eradication of multiple invasive alien species which, although challenging in planning, proved to be both successful and cost-effective. For example, a plan to eradicate five invasive alien mammal species on six islands in the archipelago of French Polynesia led to recovery of critically endangered species **(Box 5.9)**. Such management programmes on islands have also been part of large programmes focussing on social, economic and environmental objectives (**section 5.5**). On Mexican islands, a comprehensive national programme to eradicate invasive alien species and restore ecosystems, including habitat for coastal and terrestrial birds, has changed local stakeholder understanding and engagement in biosecurity policies and regulations **(Box 5.9)**. Recognizing the effectiveness of management on islands, the Global Environment Facility (GEF) has prioritized its invasive alien species funding programme towards island conservation projects (GEF, 2020).

Box 5 8 Case study: Eradication of five species of invasive alien vertebrates in the archipelago of French Polynesia.

On six islands of the archipelago of French Polynesia, a project was undertaken in 2015 to eradicate five species of invasive alien vertebrates: *Rattus exulans* (Pacific rat), *Rattus rattus* (black rat), *Felis catus* (cat), *Oryctolagus cuniculus* (rabbits) and *Capra hircus* (goats). The project was successful on five of the six islands (Pacific rats survived at one site). A management plan was developed and implemented that aimed to restore populations of the endangered *Pampusana erythroptera* (Polynesian ground dove), *Nesofregetta fuliginosa* (Polynesian storm-petrel) and *Aechmorhynchus parvirostris* (Tuamotu sandpiper), as well as other native plant and animal species. International and local conservation non-governmental organizations as well as local communities were involved from the planning phase to the execution of the management actions. Although implementation was challenging, this collective approach proved more cost-effective than if each island had been targeted individually. Effective engagement of stakeholders was key for the success of the project. The livelihood of local communities was also improved through the project (Griffiths *et al.*, 2019).

Box 5 9 Case study: National Program for Island Restoration in Mexico.

The eradication of invasive alien species was the first step of the National Program for Island Restoration in Mexico together with active ecosystem restoration for the recovery of seabirds (Bedolla-Guzmán *et al.*, 2019), biosecurity protocols (Latofski-Robles *et al.*, 2019), vegetation and soil restoration (Luna-Mendoza *et al.*, 2019), and environmental learning with local communities (Aguirre-Muñoz *et al.*, 2016). Mexican islands are extraordinarily diverse, including semi-arid islands in the Eastern Pacific Ocean; desert islands in the Gulf of California; and subtropical and tropical islands in the Pacific Ocean, the Gulf of Mexico and the Caribbean (Aguirre-Muñoz *et al.*, 2016). On these islands, 21 endemic species and subspecies of vertebrates have gone extinct in the last 100 years, and all but four of these extinctions were caused by invasive mammals see **Chapter 4**, **Box 4.4** and **section 4.3.1**). Islands were selected for ecosystem restoration and action (i.e., control or eradication of invasive alien species) based on conservation value, management efficiency, social acceptance and technical and financial feasibility (Latofski-Robles *et al.*, 2014). Initiated in 1995 at some islands, the number of target islands and species increased, and by April 2018, 60 populations of invasive alien mammals were successfully removed from 39 islands, 30 of which are now completely free of invasive alien mammals. The extent of the success of the eradication programmes can be illustrated by the numbers of populations and invasive alien species controlled across the islands: 32 populations of 12 species from 15 islands of the Pacific Ocean, 21 populations of 5 species from 18 islands of Gulf of California and 7 populations of 3 species from 6 islands of Gulf of Mexico and Caribbean. These actions are estimated to be protecting at least 147 endemic taxa of mammals, reptiles, birds and plants, as well as 227 seabird breeding colonies (Aguirre-Muñoz *et al.*, 2018).

5.3.3 Decision tree for selection of management approach

Choosing the most appropriate management objective is the first step to deciding between a pathway, species-based, site-based or ecosystem-based management approach for biological invasions, but it is not always straightforward (**section 5.2**). Objectives of a management programme for biological invasions may be aimed at economic, social or environmental outcomes or at multiple benefits. For example,

Box 5 10 **Case study: Decision tree for separating site-based versus species-based management of invasive alien plants (after Owen & Sheldon, 1996).**

The New Zealand Department of Conservation outlined a collective approach encompassing both species-based and site-based programmes for decision-making for the management of invasive alien plants. In their approach, species-based initiatives are aimed at new incursions and providing the best conservation outcome, and site-based initiatives are aimed at protecting biodiversity in terms of the collective threat and urgency for management from all alien species present, or the value of protecting a site from all invasive alien species (Timmins & Popay, 2002). Some species-based programmes are specifically aimed at protecting biodiversity (Downey, 2010). The main decision criterion for selecting sites may be the presence of one or more major invasive alien species threatening biodiversity allowing targeted control and threat abatement (Downey, 2013). To assist in making such decisions, the following decision tree may be of use **(Figure 5.14)**.

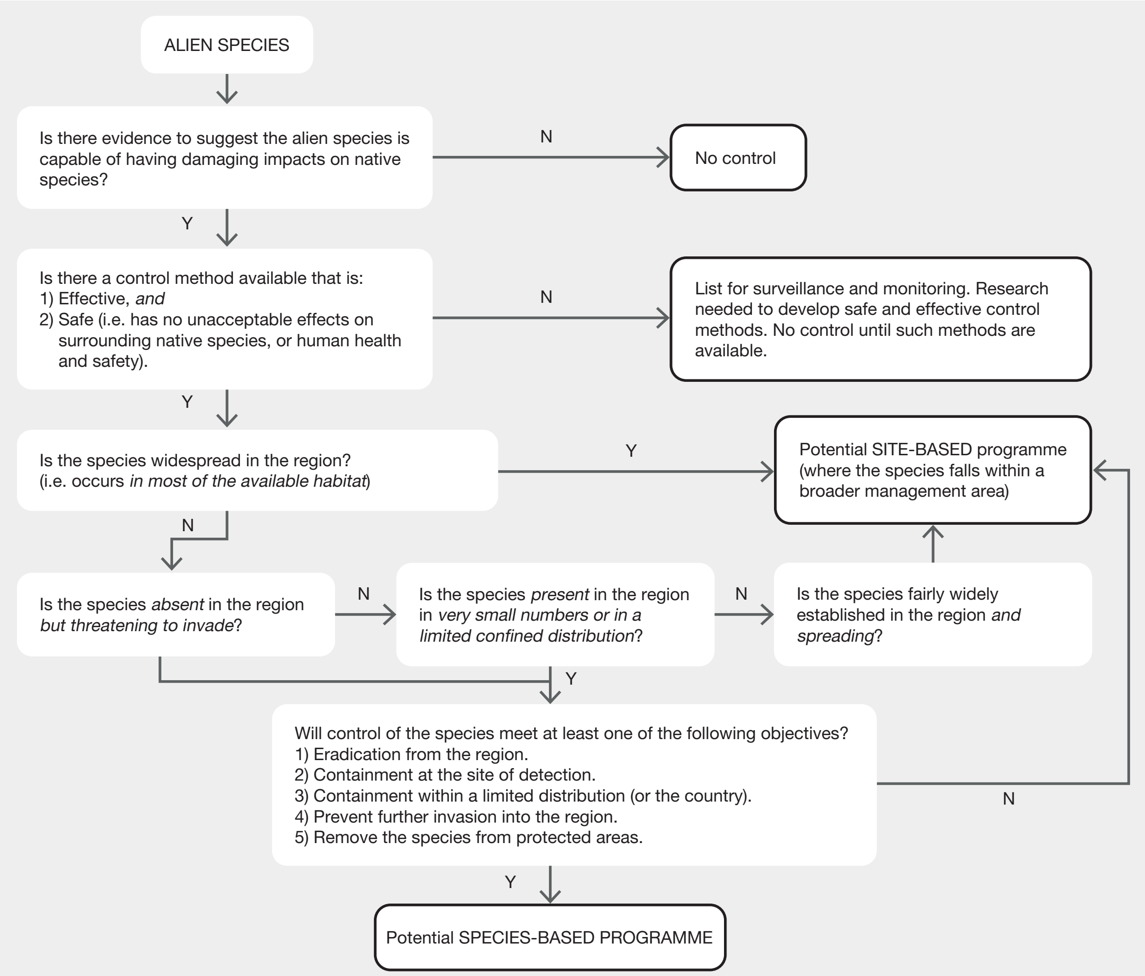


Figure 5 14 **Decision tree for choosing between site-based and species-based management of invasive alien plants.**

Double bordered rectangles are terminal nodes. Adapted from Owen & Sheldon (1996), https://caws.org.nz/old-site/awc/1996/awc199615161.pdf, under license CC BY 4.0.

programmes aimed at managing invasive alien pigs could produce benefits for biodiversity and ecosystems (nature), Indigenous Peoples and local communities' livelihoods (good quality of life and nature's contributions to people), livestock disease and property damage management (agriculture), reduced carbon emissions (mitigating climate change), or all of these (Nordberg *et al.*, 2019; Zivin *et al.*, 2000). The most cost-effective way to ensure the survival of threatened and endangered species in a region may be a collective regional planning approach to management of biological invasions (Carwardine *et al.*, 2012). It is worth considering one or all the available approaches. Confusion and considerable debate around whether and when management of biological invasions should follow a pathway, species-based or site-based management approach appears to result from poorly defined management objectives, which too often simply focus on species prevention or suppression (Downey & Sheppard, 2006). For example, the Australian Weeds of National Significance programme (Thorp & Lynch, 2000) and associated invasive alien plant classical biological control programmes (Downey & Sheppard, 2006) target the highest priority invasive alien plant species. Thus, it is important that the aim of any species-based or site-based invasive alien species initiative be clearly articulated. **Box 5.10** describes a decision-support system developed to help decide when to undertake pathway, species-based or site-based management for invasive alien plants.

5.4 REVIEW OF KEY DATABASES, MANAGEMENT TOOLS AND TECHNOLOGIES FOR BIOLOGICAL INVASIONS

From countries to communities, cost-effective solutions for managing pathways, invasive alien species and invaded sites and ecosystems are needed to prevent increasing economic, environmental and social impacts (Ricciardi *et al.*, 2017). A wide range and increasing number of data sources, tools and technologies support actions towards a) prevention, preparedness, surveillance, detection and monitoring of pathways, b) species-based eradication, containment and management and c) site-based and ecosystem-based management to protect key biodiversity assets to build resilience to further invasion. This section provides explanatory information on databases, tools, technologies, platforms and approaches to support their adoption and use, and the context in which they can be used in the management of biological invasions. Limitations, challenges, advantages, and disadvantages of these are also covered in either the body of the section or in the **Supplementary materials 5.3** to **5.8**. These key databases, management tools and technologies for biological invasions are categorized under a) databases, b) surveillance, detection and diagnostics and c) intervention technologies. For the last two categories, the tools and technologies are grouped in the context of managing pathways, species and sites. In each of these categories, stakeholder engagement frameworks and decision-support tools have been covered in **section 5.2**. Some of the latest technologies with significant potential, but not yet applied in the context of biological invasions, are also briefly discussed. The rapid development of novel technologies and approaches has produced tools capable of massively improving management of biological invasions. However, understanding how to facilitate context specific adoption, application and operationalization of such technologies in a policy and community acceptability context is lagging behind (Burke *et al.*, 2005; Stilgoe *et al.*, 2020; van Rees *et al.*, 2022; **Chapter 6**, **section 6.3.3.4**).

Summary tables are provided to frame each tool, technology, approach or platform in terms of the aspects of management they address, their relevance to different types of invasive alien species and the temporal and spatial scale of their application.

5.4.1 Relevant databases for management of biological invasions

Accurate and real-time publicly accessible geospatial databases of invasive alien species provide considerable value for underpinning management (e.g., Seebens *et al.*, 2017; **Chapter 2**, **section 2.1.4**; **Chapter 6**, **sections 6.6.1** and **6.6.2**). There remain considerable sensitivity issues for invasive alien species important for trade and market access, but most species data is on publicly available platforms (e.g., GBIF). Big-data analytics are proving increasingly valuable for understanding and managing priority invasive alien species issues (Hay *et al.*, 2013; Bennett, 2015), but are undertaken by dedicated data analysis groups in academia, governments and industry which does not help social engagement (Lawrence, 2006). Such databases are relevant for analysing species distribution and abundance, outbreak management and also capture of management activities and their effectiveness. Globally, documentation and data on management and control costs are very limited in terms of final outcomes on nature and nature's contributions to people. One recent significant database on impacts is the InvaCost database (**Chapter 4**, **Box 4.13**). Global, regional and taxon specific databases relevant for invasive alien species management are listed in **Table 5.4** for a range of information types. Important databases for management of biological invasions include the Database of Island Invasive Species Eradications (DIISE), Biological Control of Weeds – a world catalogue of agents and their target weeds, and BIOCAT for invertebrate pest biocontrol. The IUCN Red List of Threatened Species, which currently has assessed the risk of extinction for 142,577 species, uses a hierarchical classification scheme to record drivers of species decline, including threats from invasive alien species (Salafsky *et al.*, 2008).

All these databases are subject to some degree of geographical and taxonomic or sampling bias (Yesson *et al.*, 2007). Data sharing and data integration work as long as international data standards are defined and followed. Achieving this in developing countries remains a challenge and it is important to have sustained support to ensure databases are not just a snapshot in time. In this context the CBD invasive alien species *ad hoc* technical expert group made the following observations:

- Ensure open access to databases, knowledge sources and analytical data tools *via* national and international data portals.
- Improve databases for marine, invertebrates, microorganisms and fungi and collect and integrate deoxyribonucleic acid (DNA) sequence data into existing databases, where possible.
- Develop internationally agreed data standards to facilitate data sharing.
- Develop invasive alien species filters for existing species databases (e.g., ECOLEX & FAOLEX; **Table 5.4**)
- Collectively collate data and knowledge on best practice invasive alien species policy and regulatory, voluntary codes of conduct across sectors across and international agencies and conventions.

Table 5 4 **Information components including description and importance of the information for documenting and managing biological invasions (reason) of existing invasive alien species databases (data and knowledge products) relevant for planning and implementation of management.**

Websites are provided at the first mention of each database (see **Chapter 2** for databases relevant for status and trends and **Chapter 6**, **section 6.6.3** for databases supporting policy options). Identified gaps identified within the data and knowledge products are also given. Adapted from CBD (2019).

Fields	Description	Database purpose	Examples of data and knowledge products	Identified gaps
Taxonomy	Scientific name, higher taxonomy, synonyms, common names	Name consistency & locating specimens	• GBIF – https://www.gbif.org/ • World Register of Introduced Marine Species – http://www.marinespecies.org/introduced/ • FishBase – https://fishbase.org/ • Plant List – http://www.theplantlist.org/ • The Reptile Database – http://www.reptile-database.org/ • AlgaeBase – https://www.algaebase.org/ • IUCN Red List of Threatened Species – https://www.iucnredlist.org/	Underrepresented biomes and taxa
Identification	Identification guides, diagnostic tools	Correct identification, Early Detection	• iNaturalist – https://www.inaturalist.org • Lucidcentral – https://www.lucidcentral.org • Antweb – a comprehensive diagnostic tool for ants – http://antweb.org/ • Plant net – https://plantnet.rbgsyd.nsw.gov.au/ • eBird – https://ebird.org/home • BioNET – EAFRINET – https://keys.lucidcentral.org/keys/v3/eafrinet/plants.htm • Portaleei Latin America – http://portaleei.fcien.edu.uy/	
Ecology	Including habitat, species interactions (e.g., host species)	Management Risk assessment	• Global Invasive Species Database (GISD) – http://www.iucngisd.org/gisd • Centre for Agriculture and Bioscience International Invasive Species Compendium – https://www.cabi.org/isc • FishBase • National invasive alien species databases – http://www.inbiar.uns.edu.ar/; http://bd.institutohorus.org.br; https://caribbeaninvasives.org; https://sieei.udelar.edu.uy; https://guyra.org.py; https://invasoras.biodiversidad.gob.ec	

Table 5 4

Fields	Description	Database purpose	Examples of data and knowledge products	Identified gaps
Spatial data	Distribution, native and introduced range, occurrence	Origin, Management, Risk assessment	• Global Invasive Species Database • Global Register of Introduced and Invasive Species (GRIIS) – http://www.griis.org/ (Pagad *et al.,* 2018, 2022b, 2022a) {Table 5.4} • Centre for Agriculture and Bioscience International Invasive Species Compendium • FishBase • Global Naturalized Alien Flora (GloNAF) – https://glonaf.org • Global Avian Invasions Atlas – https://doi.org/10.6084/m9.figshare.4234850.v1 • SeaLifeBase – https://www.sealifebase.ca • WOAH – https://www.woah.org/en/what-we-do/animal-health-and-welfare/disease-data-collection/world-animal-health-information-system/ • European Alien Species Information Network – https://easin.jrc.ec.europa.eu/easin/# • Pacific Islands Ecosystems at Risk – http://www.hear.org/pier/ • Species observations for the United States and Territories – https://www.gbif.us • Atlas of Living Australia. Analytic software platforms, extensive and open source – www.ala.org.au • National invasive alien species databases • Biomodelos – Biomodels of potential distribution maps and invasive species fauna and flora in Colombia – http://biomodelos.humboldt.org.co/en • International Union for Conservation of Nature Red List of Threatened Species • Regional plant protection organizations – https://www.ippc.int/en/external-cooperation/regional-plant-protection-organizations/	
Status and Provenance	Invasive alien species status in introduced range including abundance, occurrence (extent of spread) and invasiveness	Origin, Prioritization and Management Prioritization	• Global Invasive Species Database • Global Register of Introduced and Invasive Species • Centre for Agriculture and Bioscience International Invasive Species Compendium • FishBase • European Alien Species Information Network • Pacific Islands Ecosystems at Risk • World Register of Introduced Marine Species • SeaLifeBase – https://www.sealifebase.ca/ • WOAH World Animal Health Information System – disease status • National invasive alien species databases	
Primary and secondary pathways	Intentional or unintentional Pathways of introduction and spread	Biosecurity Management	• Global Invasive Species Database • Global Register of Introduced and Invasive Species • Centre for Agriculture and Bioscience International Invasive Species Compendium • FishBase • European Alien Species Information Network • Pacific Islands Ecosystems at Risk • World Register of Introduced Marine Species • Database on Introductions of Aquatic Species • IPPC Documentation on ISPM – https://www.ippc.int/en/core-activities/standards-setting/ispms/ • National invasive alien species databases http://www.inbiar.uns.edu.ar/	Secondary pathways classification inconsistent or missing

Table 5 4

Fields	Description	Database purpose	Examples of data and knowledge products	Identified gaps
Monitoring and surveillance	Data from multiple sources in a real time	Early Detection	• Early Detection and Distribution Mapping System – https://www.eddmaps.org/	
Impact	Environmental and socio-economic impact, mechanisms of impact, outcomes of these impacts and ecosystem services impacted	Risk assessment Policy Management	• Global Invasive Species Database • Global Register of Introduced and Invasive Species • Centre for Agriculture and Bioscience International Invasive Species Compendium • InvaCost database – https://figshare.com/articles/dataset/InvaCost_References_and_description_of_economic_cost_estimates_associated_with_biological_invasions_worldwide_/12668570/4 • Millennium ecosystem assessment – https://www.millenniumassessment.org • IUCN Red List of Threatened Species – https://www.iucnredlist.org/resources/threat-classification-scheme • FishBase	No transparent, standardized way to report on impacts
Risk assessments	Developed risk assessments with outcomes	Management	• Global Invasive Species Database • Pacific Islands Ecosystems at Risk • Environmental Impact Classification of Alien Taxa (EICAT) and the Socio-Economic Impact Classification for Alien Taxa (SEICAT) • Global Compendium of Weeds – http://www.hear.org/gcw/ • East and South European Network for Invasive Alien Species – www.esenias.org • Pacific Invasive Ants Toolkit – http://www.piat.org.nz/ • National invasive alien species databases	
Policy response	Legislations enacted, regulations, voluntary codes of conduct	Policy Management	• ECOLEX – https://www.ecolex.org • FAOLEX – fao.org/faolex/en/ • InforMEA – United Nations Information Portal on Multilateral Agreements – https://www.informea.org • EU Regulations – https://ec.europa.eu/environment/nature/invasivealien/index_en.htm	Databases not searchable for invasive alien species
Eradication	Successes	Management	• DIISE – http://diise.islandconservation.org/ • Global Eradication and Response Database – http://b3.net.nz/gerda/ • National invasive alien species databases	
Control	Management practices, failure, best practices, biocontrol	Management	• Pacific Islands Ecosystems at Risk • Database of introductions of insect biological control agents for the control of insect pests (Cock *et al.*, 2016) {Table 5.4} • Biological Control of Weeds. A world catalogue of agents and their target weeds – https://www.ibiocontrol.org/ • iMapInvasives – sharing information for strategic management – https://www.imapinvasives.org • Centre for Agriculture and Bioscience International Invasive Species Compendium • Pacific Invasive Ant Toolkit • Caribbean Invasive Alien Species Network – https://caribbeaninvasives.org/ • Database of Island Invasive Species Eradications • Global Eradication and Response Database • Early Detection and Distribution Mapping System • East and South European Network for Invasive Alien Species • National invasive alien species databases	No standardized way to report on management outcomes

5.4.2 Surveillance, detection and diagnostics supporting prevention and preparedness

There are a range of tools and technologies for surveillance, early detection and monitoring of invasive alien species including measuring the effectiveness of management actions. These include remote sensing (satellite and aerial imagery, drones, under water remote vehicles, camera traps etc.), sensor networks and crowd sourcing and the traditional use of trained detector dogs (e.g., Browne *et al.*, 2006). These tools are becoming increasingly cost-effective for early detection (**section 5.1**). Early detection also needs effective species-based diagnostics tools, not all of which are based on taxonomy or morphological characteristics as described here. Technology adoption is heavily driven by cheaper price differentiation under novel business models. This is what is frequently termed "disruptive technologies".

5.4.2.1 Pathway surveillance tools and technologies

a) Digital data mining – crowdsourcing general surveillance

Citizen surveillance, through crowdsourcing and data-mining or web scraping, social media and other data streams filtering on invasive alien species content, can be used as a cost-effective complementary form of general surveillance supporting species-based risk assessment (Grossel *et al.*, 2017; Lyon, 2010; Welvaert & Caley, 2016). Data mining is extracting information from large databases. Resources scanned can include internet search engines, Really Simple Syndication (RSS) feeds and Twitter, which often contain invasive alien species photographic, taxonomic or detection-based content. Searches can be targeted at specific species or can be more general (e.g., symptoms/impacts) and can include other terms such as climate and land use change. Software exists (e.g., International Biosecurity Intelligence System (IBIS)) which can automatically search the internet daily looking for invasive alien species reports, grey literature, articles from relevant journals and any other articles or comments, thereby generating invasive alien species intelligence. Crowdsourcing surveillance can include early warning, mapping, eradication, containment, understanding real-time impacts, proof of area wide pest/disease freedom (for trade purposes) and knowledge sharing. Once an article is found, third-party web services such as AlchemyAPI and GeoNames can be used to extract information from the article such as the title, text, author, language and locations. This approach accesses citizens as surveillance agents, as "eyes and ears" over large areas. The key difficulty lies in delineating real incursion events from background "noise". Costs are limited to crowdsourcing system development, hardware and software maintenance. Systems exist for biosecurity (e.g., IBIS; Grossel *et al.*, 2017), animal and public health diseases (e.g., Program for Monitoring Emerging Diseases (ProMED; M. Carrion & Madoff, 2017), linked to EpiSPIDER (Tolentino *et al.*, 2007); BioCaster (Collier *et al.*, 2008). The multiple global and jurisdictional coronavirus disease 2019 (COVID-19) dynamic online case number dashboards are other examples of real-time automated invasion surveillance data feeds.

b) Sensor networks and smart traps

An emerging cost-effective approach to passive surveillance is through the use of sensor networks (Farouk & Zhen, 2019; Rundel *et al.*, 2009) and mobile smart traps (Potamitis & Rigakis, 2015). Wireless sensor networks generally consist of a number of different sensors connected wirelessly, that typically collect audio, image and body temperature observations produced by monitored targets, and use machine learning and pattern recognition algorithms to identify targets of interest automatically from these observations. Such networks can provide effective methods for small-scale continuous monitoring applications. They provide multiple observations operating independently over long time periods. The infrastructure deployment and maintenance costs, however means spatial coverage is limited (Preti *et al.*, 2021). Key advantages include low power allowing for the deployment of many and varied sensors across a landscape, continuous data streams provide real-time data transferred *via* the mobile network even if accessible by only a few sensor nodes. Such systems can be applied in terrestrial, aquatic (Kong *et al.*, 2005) or aerial (Kgori *et al.*, 2006) settings. Attaching sensors to mobile objects, such as domestic and wild animals, has the potential to greatly extend the spatial coverage of fixed sensors (Duda *et al.*, 2018). Sensor networks can be deployed in tracking invasive alien species movement and activities, invasive alien species in lakes, rivers, or reefs, as well as on birds, flying foxes or similar (Jurdak *et al.*, 2013; K. Li *et al.*, 2014). Networked mobile suction traps or smart lure traps can also be cost-effective and be used at high- risk sites such as ports of entry or at jurisdictional borders to monitor pest movement pathways (Harrington *et al.*, 2012). Trap contents can potentially be analysed *via* metabarcoding environmental DNA (Lagos-Kutz *et al.*, 2020; **section 5.4.2.2h**; **Glossary**).

Over the past decades, wireless sensor networks (including lightweight telemetric tags) have been deployed successfully in a number of invasive alien species contexts reviewed by Jurdak *et al.* (2015), including to detect insect pests (López *et al.*, 2012), invasive alien vertebrates (Fleming *et al.*, 2014), invasive frogs (Hu *et al.*, 2009), fish (Jurdak *et al.*, 2015; Kottege *et al.*, 2012) and flying foxes (Sommer *et al.*, 2016). Infrared cameras have also been used for livestock biosecurity to collect body temperatures of cattle as a sign of disease infection (Rainwater-Lovett *et al.*, 2009). Low-power image-sensor networks have been used to detect and classify insect pests (Jurdak *et al.*, 2015) and invasive alien vertebrates. See **Supplementary material 5.2** for further details.

c) Screening technologies

X-ray screening devices are now in operation at most airports and have been a standard technology of biosecurity operations at borders for a number of years (Whyte, 2006). Their quality as a screen technology is variable due to the cost of both software and proper training of personnel. Human behaviour can also compromise effectiveness, for example proper fluid detection systems are often circumvented by staff because they cause frequent machine errors. Next generation 3D x-ray machines are however much more sophisticated and are being installed at airports and postal mail sorting centres in some countries (e.g., Australia and New Zealand; Australian Government, 2021a). Digital triage of these types of images will increasingly be run autonomously with machine learning algorithms trained to risk profiling key indicators of suspect material (Marturana *et al.*, 2015). This could potentially lead to autonomous screening of luggage or postal mail triaging suspect items for human inspections. Similar systems are also under development for scanning shipping containers (C. H. Lim *et al.*, 2021).

d) Environmental DNA

All organisms leave a genetic trace of themselves within their environment and there are multiple ways of sampling and analysing this environmental DNA for species detection (C. I. M. Adams *et al.*, 2019; Herder *et al.*, 2014; Truelove *et al.*, 2022) including for invasive alien species (Rees *et al.*, 2014; Bylemans *et al.*, 2019). When applied within pathway or ecosystem surveillance, control or eradication programmes (Carim *et al.*, 2020), environmental DNA analysis provides a sensitive and efficient means to detect the presence of a particular or multiple species and is applicable to all organisms including microbes that exceed the sensitivities of conventional observational monitoring (Furlan *et al.*, 2019), particularly for situations where other novel detection technologies are not applicable (Bylemans *et al.*, 2019; **Chapter 6**, **section 6.6.1.2**; **Box 6.19**). Environmental DNA analysis has the potential to be applied across most fields of invasive alien species, environmental research and land management, particularly for aquatic and marine invasive alien species (C. Abbott *et al.*, 2021) as many conventional and novel methods for detection do not work well (if at all) in these environments. Environmental DNA can be applied, for example, to identify species within aquatic or marine environments using polymerase chain reaction (PCR, see **Glossary**) tools or for whole community assessments using metabarcoding (Rees *et al.*, 2014; Bylemans *et al.*, 2019). Portable environmental DNA PCR units are also increasingly available. More recently studies are demonstrating the capacity to also capture and analyse aerial environmental DNA (Banchi *et al.*, 2018). Cross-phyla studies combining environmental DNA metabarcoding with taxonomy and population genetics is also being used to detect new introductions of species and genotypes (Holman *et al.*, 2019). Care is needed in the interpretation of results, as the presence of a species' DNA does not necessarily mean that live individuals are also present and cannot determine where in the sampled environment the species are/were. Environmental DNA density may not be correlated to species abundance as DNA can accumulate or persist in certain situations or be degraded or lost in others. Some target species have a strong seasonal release/availability of environmental DNA (e.g., crabs and other marine fauna), which complicates detection. Sampling design may substantially influence sensitivity (Furlan *et al.*, 2016; Hinlo *et al.*, 2017) and abundance biases (Furlan *et al.*, 2018). Broad-scale environmental DNA sampling is relatively of low cost, and data acquisition and extraction can be streamlined. Environmental DNA sampling is non-invasive and non-destructive to sensitive environments.

Environmental DNA is consistently used to demonstrate absence of selected invasive alien species, and could be combined with other approaches where possible. To help manage the issue of live and dead species in samples, a refined system is being developed where only DNA from whole cells in the sample are collected. These sorts of approaches are rapidly being adopted in relevant biosecurity monitoring programmes in for example Australia and New Zealand and can also be usefully combined with citizen science (E. R. Larson *et al.*, 2020).

Sampling for environmental DNA analysis can be coupled to unmanned vehicles, including aerial, ground-based and aquatic or marine vehicles with sample analysis undertaken autonomously. In some scenarios, this combined approach might effectively automate the process of surveillance and enable far greater penetration of inaccessible landscapes or environments. See **Supplementary material 5.2** for further details.

e) Sentinel surveillance and monitoring

Sentinel surveillance has been developed for early detection, surveillance and monitoring of invasive alien disease incursions. It involves sampling from a sentinel species and may be configured to identify a single or a range of invasive alien species. Targets commonly include infectious animal diseases (Batista *et al.*, 2012; McCluskey & Salman, 2003) and plant pests and pathogens (Kenis *et al.*, 2018). Animal disease examples include West Nile virus using chickens and mosquitos (Reisen *et al.*, 2004), cattle for bluetongue disease in sheep (Elbers *et al.*, 2008) and bovine ephemeral fever virus in cattle (St George, 1985) or bovine tuberculosis using pigs, badgers and cattle (McInerney *et al.*, 1995;

Murphy *et al.*, 2011). Relatively less is known about the sensitivity, practicality and other characteristics of sentinel disease surveillance using wild species providing for sentinel surveillance. In one example in Estonia and Latvia, regular testing (and aerial vaccine bait dropping) is undertaken for wild animal rabies (e.g., raccoon dog) (Holmala & Kauhala, 2006). The monitoring of wildlife to understand the incidence and risk of particularly zoonotic diseases is likely to increase post COVID-19 (Latinne *et al.*, 2020).

The principles and value of sentinel surveillance are also acknowledged by WOAH, and can be utlized in the One Health approach (**Glossary**; **Chapters 1** and **6**). There are no existing regulatory precedents for the use of wildlife species in sentinel surveillance. The increasing development and use of novel point-of-detection rapid diagnostics creates an opportunity for sentinel surveillance to be augmented (**section 5.4.3.2**). In addition to sentinel plant surveillance nurseries for plant pests and diseases[5] (Eschen *et al.*, 2019; Kenis *et al.*, 2018), sentinel sites (**Glossary**) have also been proposed for monitoring for new invasive alien plant invasions in key disturbed locations close to points of entry (T. J. Mason *et al.*, 2005), such as sites where there are generally high records of alien plant naturalizations such as waste dumps (Clements & Foster, 1994). See **Supplementary material 5.2** for further details.

5.4.2.2 Species-based surveillance, detection and diagnostics tools and technologies

a) Citizen science – surveillance data input portals and diagnostics platform

Citizen science reporting of invasive alien species presence and or impacts through data portals or hotlines (explicitly dedicated telephone numbers) is now widely recognized as a very effective form of active general surveillance (Crall *et al.*, 2011; Welvaert & Caley, 2016; E. R. Larson *et al.*, 2020; Johnson *et al.*, 2020; Aceves-Bueno *et al.*, 2017; **Chapter 1**, **section 1.6.8**, **Box 1.15**; **Chapter 6**, **section 6.6.2.1**). Large scale citizen-science is being used to monitor disease-carrying mosquitos in southern Europe (Mosquito Alert, 2021). New Zealand has effectively directly targeted its population in biosecurity campaigns as the "eyes and ears" of their national biosecurity system. Reporting of species sightings is through smart phone apps and other online biological recording or reporting platforms such as *iNaturalist* and national reporting hotlines often supported by online taxonomic tools and resources. Many local and open-source adaptable biodiversity, pest and disease reporting apps exist now in many countries (e.g., BioCollect hubs on the Atlas of Living Australia). Portals can be regionally specific, habitat or biome specific (e.g., RedMap in Australia or European Alien Species Information Network in Europe) or organism type specific. In Canada, citizen science is being used for marine invasive alien species surveillance (Delaney *et al.*, 2008). A purpose-built regionalized system is the "Invaders of Texas program" (Gallo & Waitt, 2011).

Limitations around relevance only to relatively easily observable and identifiable species are being addressed by automated off the shelf digital platforms (Schmidt-Lebuhn & Norton, 2017; Wäldchen *et al.*, 2018; **section 5.4.3.2c**). Citizen science portals are less effective for species generally requiring laboratory-based diagnostics (e.g., for micro-organisms and diseases) but not always (see AshTag App. in the United Kingdom for ash dieback). There are also privacy concerns if publicly searchable data repositories can identify landowners legally responsible for invasive alien species on their properties or include unvalidated records of potentially trade-sensitive species of biosecurity relevance. Citizen science reporting is also relevant beyond species distribution and abundance to outbreak management. Data input portals can also be used to capture management activities and their effectiveness.

b) Earth observation – remote sensing detection

Remote sensing is an important tool supporting invasive alien species surveillance and monitoring (C. Joshi *et al.*, 2004), eradication, containment and widespread management (Walsh, 2018). The growing availability and adaptability of remote sensing could make large-scale eradication programmes cost-effective.

Earth observation data from satellites and manned and unmanned aerial systems allows rapid, large-scale and repeatable assessment of areas inaccessible to ground surveys (Pettorelli *et al.*, 2014; Royimani *et al.*, 2019). Artificial Intelligence algorithms allow unmanned aerial systems to return to suspected detections and take repeat images from a number of angles for confirmation (Gonzalez *et al.*, 2016). The capacity for real-time analysis and information delivery is evolving as computing power improves. Ongoing technological and analytical advancements continue to improve both sensitivity and cost-effectiveness. The availability of specific algorithms is a key limitation, as these would need to be developed for each application. Data generally come from two types of sensors; the passive sensors (such as multispectral, hyperspectral, or thermal) and active sensors, such as radar or light detection and ranging (Lidar) using laser pulses to measure reflection times in space, and providing detailed information on vegetation structure and understory (Dash *et al.*, 2019).

While advancements are in progress, satellite imagery is still limited by the resolution, time, frequency of overhead passage and spatial detail which for passive sensors is

5. https://www.plantsentinel.org

limited to cloud-free periods. Satellite imagery currently complements aerial systems, however, the deployment of small low orbital satellite constellations could, in addition to providing capacity for the internet-of-things to remote areas, help specific invasive alien species recognition and monitoring (Schnase *et al.*, 2002). With the growing availability of free or low-cost higher spatial resolution operational satellites such as Sentinel and CubeSats, larger infestations can be monitored; however, small patches and individual plants are still impossible to detect on coarser resolution data, being especially true for the highly heterogeneous landscapes where the occurrence of invasive alien species plant populations is rather patchy (Perroy *et al.*, 2017). While conventional manned aircraft (including helicopters) are still widely used, advances in miniaturization of imagery platforms on unmanned aerial systems is leading to replacement. Very high spatial resolution and flexibility of unmanned aerial systems holds great potential to support targeted monitoring, identify priorities for management and assist eradication and suppression of invasions (Müllerová *et al.*, 2017). Remote sensing from drones (**Supplementary material 5.3** for their limitations) can target specific tasks in time and across limited space, but at finer detection resolutions, working below clouds and avoiding excessive wind speeds. Multiple sensors and high resolution massively increases the data storage and complexity, analytics and processing time (Müllerová, 2019). Sensors can target invasive alien species directly using a specific optical signature or can detect their presence indirectly through methods such as rapid change in a landscape parameter over time caused by their spread. See **Supplementary material 5.3** for further details.

Examples of the use of remote sensing employing unmanned aerial systems imagery in invasive alien plant detection and active management include multiple life forms from the wet and dry tropics to Mediterranean and temperate regions (Elkind *et al.*, 2019; Hill *et al.*, 2017; Lehmann *et al.*, 2017; Lopatin *et al.*, 2019). In the European Union radar is being applied to track Asian hornets (LIFE STOPVESPA, 2021). Remote sensing can also be used well beyond mapping pest and disease distribution, density

Box 5 11 Case study: *Solenopsis invicta* (red imported fire ant) eradication.

In Brisbane (Australia), the eradication of fire ant is dependent on airborne imagery. A camera is mounted beneath a helicopter, which flies over the target area at a height of 150m. Images are captured in three separate frequency ranges: visible, near infrared and thermal. These are then processed in parallel to identify objects that may be fire ant nests which are then destroyed by direct injection **(Figure 5.15)**. As the size and weight of the camera decreases, there is potential to replicate the approach using unmanned aerial systems. This would be significantly cheaper than the use of helicopters, and would allow for significantly more surveillance. Red imported fire ant has been eradicated five of the six times they have established in Australia and these and other tramp ants have been intercepted a total of more than 225 times in recent years. Airborne imaging is a critical technology in this context and may also be applicable to other tramp ant eradication programmes (Hoffmann *et al.*, 2016).

Figure 5 15 **The manual application of a chemical treatment of *Solenopsis invicta* (red imported fire ant) in Brisbane Australia as part of an eradication campaign.**

Photo credit: The State of Queensland – Department of Agriculture and Fisheries 2019 – under license CC BY 4.0.

and damage. GIS and remote sensing technologies are highly advanced in mapping invasive alien species. Indirect remote sensing has been used to detect and map cryptic understory invasive alien species in forests (C. Joshi *et al.*, 2006). Remote sensing is also being used to detect diseases before symptoms occur (e.g., *Xylella fastidiosa* (Pierce's disease of grapevines)) and for assessing habitat suitability for invasive alien species (e.g., Zarco-Tejada *et al.*, 2018). Thermal imagery has been used to replace surveys of invasive alien vertebrates at low density in open habitats and inaccessible locations (Amstrup *et al.*, 2004; Storm *et al.*, 2011) and for monitoring tramp ants **(Box 5.11)**.

c) Automated image-based diagnostics

Machine learning or artificial intelligence is being developed to contribute to the automatic identification and diagnosis of plant pests and diseases (Dawei *et al.*, 2019; Jia & Gao, 2020). Automated image library based digital diagnostics platforms for invasive alien plants, invertebrates and pathogens that can be used on mobile devices are supporting biosecurity inspections and citizen science (Chen *et al.*, 2021; Schmidt-Lebuhn & Norton, 2017; Wäldchen *et al.*, 2018). Artificial intelligence or machine learning algorithms combined with image-processing-based species-recognition software provide automated triage of identification likelihood quickly and easily either back to the user or for uploaded images being signalled to an expert in the context of possible high priority targets. This reduces the effort in searching for false positive notifications which weaken analyses of species distributions (Mo *et al.*, 2017). Similar systems are already being used in public health as a technological tool enabling rapid screening with high accuracy (Chowdhury *et al.*, 2020; van de Kant *et al.*, 2012). The learning architectures and algorithms are becoming highly sophisticated with the development of the convolutional neural network, a deep learning network important in image recognition (Luaibi *et al.*, 2021). For example, the architecture was useful in classifying citrus diseases and insect damage from leaf images to best accuracy with data augmentation to a level of 97.9 per cent (Luaibi *et al.*, 2021) and chicken sound convolutional neural networks were used to differentiate Avian influenza in poultry (Cuan *et al.*, 2020).

d) Volatile detection technologies

Volatile detection technologies can be configured to identify any or multiple targets with a unique volatile profile, or footprint (Cui *et al.*, 2018). These technologies have relevance for a) detection of terrestrial invasive alien species offshore, at port of entry or onshore, b) detection of plants and animals infested with pests or diseases supporting conventional disease diagnostics (Knobloch *et al.*, 2009; Laothawornkitkul *et al.*, 2008), c) screening of international goods, postal mail, travellers, luggage, cargo, containers, ships and aircraft (e.g., Staples & Viswanathan, 2008) and d) improving invasive alien species traps and lures (e.g., Sweeney *et al.*, 2004).

Trained detector dogs are the conventional approach in most situations for a wide range of invasive alien species threats (A. Y. Moser *et al.*, 2020), but require direct human support and high training and maintenance costs. Many hand-held point-of-use detections systems have been developed for volatile detection, including miniaturized portable gas chromatography mass spectrometry and Fourier-transform infrared spectroscopy (FTIR), array-based sensors such as electronic noses and biosensors (Berna *et al.*, 2009; A. D. Wilson, 2017). Key advantages of the volatile detection technologies include reliable detection from small samples, use for single or multiple targets, and low search effort and faster potentially automated screening. International approval may be required where proposed for use to replace other tests agreed under trade conventions (e.g., WTO) to demonstrate equivalent sensitivity and specificity. See **Supplementary material 5.3** for further details including on advantages and disadvantages of the different volatile detector technologies.

e) Pheromone and semiochemical lures

In addition to the traditional use of pheromone lure traps for the management of established invasive alien species (mostly invertebrates as pests largely in horticulture (El-Sayed *et al.*, 2006), such lures can and are being very effectively used in surveillance and detection (Augustin *et al.*, 2012) and delimitation and spread in eradication and containment programs (Brockerhoff *et al.*, 2010; Suckling *et al.*, 2014) of newly established alien species. The general approach is the application of chemical ecology to identify and then manufacture specific pheromones (volatile chemicals used by species e.g., in sexual attraction) or other semiochemicals that attract the target invasive alien species. These chemicals are then distributed in a lure or bait (usually made of material that can store and also slowly release the chemical) inside a trap, such that the target species is attracted too and trapped for detection and identification. The distribution of a network of lures in the area where detection is considered most likely provides a detection system (generally attracting males of the species to a female sex pheromone) that is very useful for demonstrating whether species (in numbers capable or reproduction) are present/absent and to some degree determining species abundance (R. A. Hayes *et al.*, 2016; Suckling, 2015).

f) Acoustic/ultra-sound sensors

Ultra-sound and acoustic surveillance devices can be used to detect target invasive alien animals (Demertzis *et al.*, 2017; Jurdak *et al.*, 2015). Performance is affected by many factors including the sensor type and frequency used, the size and behaviour of the target, the distance between the target and the sensor, the sampling time and the structure of the substrate where the target is found. Some invasive alien animals produce diagnostic sounds and there are a range of acoustic, sound and vibration sensors commercially available. Beyond this, investment may be needed to determine the suitability acoustics for different invasive alien animals in different contexts and to build the digital platforms for data collection and analytics. For marine species identification, an advanced machine hearing framework can be applied to target invasive alien species based on the sound they produce. The hearing framework uses two effective machine learning algorithms, the online sequential multilayer and the graph regularized extreme learning machine autoencoder that provides a higher level of generalization (Demertzis *et al.*, 2018). Checking whether the recognized species is native to its locality or not is carried out by using global positioning systems (Demertzis *et al.*, 2018). There may also be some utility for niche applications such as detection of invasive alien insects in containers, which would be supported by a large body of research on stored grain pest detection and pests in timber (Zahid *et al.*, 2012). Bioacoustic sensor networks have been developed and deployed to detect invasive alien frogs (*Rhinella marina* (cane toad); Hu *et al.*, 2009) and invasive fish (Kottege *et al.*, 2012) from their calls.

g) Point of Care / Lab on a chip, rapid test diagnostics

Handheld rapid diagnostic test platform Point-of-Care (PoC) diagnostics and Lab on a chip (LoC) are becoming increasingly common to diagnose human (Riccò *et al.*, 2020), animal (Gattani *et al.*, 2019) and plant diseases (Lau & Botella, 2017). The global spread of African swine fever and associated rapid diagnostic platform development illustrates their value for tracking invasive alien diseases (Ye *et al.*, 2019). Most handheld systems are designed to detect specific gene sequences or proteins. PoC options for many pathogens rely on immunoassays in dipstick, lateral flow devices or increasingly microfluidic platforms (Weng *et al.*, 2019). These PoC and LoC are new diagnostic kits that generally need to be rigorously tested for sensitivity, selectivity, or performance. While microfluidic options lend themselves to multiplexing to diagnose multiple pathogens, a true combinatorial approach to diagnostics would be required to deliver a generic diagnostic platform. Currently, to detect a pathogen a specific sensor is required and to detect two different pathogens two different sensors are required and so forth. A combinatorial approach with sensors with broad but overlapping detection of pathogens would allow a multiplexed diagnostic that could sense more pathogens than the number of sensors it contains. Pooling samples will depend entirely on the sensitivity of the PoC diagnostic as pooling may dilute the target analyte. There are many lateral flow devices available for diagnosis of particular pathogens, however, there are little data comparing the performance, sensitivity, and specificity of the tests compared to standard laboratory diagnostic assays. International approval may be required where proposed for use to replace other tests agreed under trade conventions (e.g., WTO) to demonstrate equivalent sensitivity and specificity. Next generation sequencing (NGS) may be able to detect the presence of many pathogens and DNA tests are more sensitive than immunoassays. Currently, NGS technology is not available at the PoC as it still requires multiple steps for sample preparation, amplification, sequencing and in-depth analysis. NGS technology improvements are emerging at a faster rate as it is a hot topic of research to enable human health diagnostic improvements. A large research effort would be needed to target next generation sequencing (NGS) to plant pathogens or other targets. The next level of automation is to deploy LoC technology that is capable of high-throughput screening for the pathogens of interest, utilizing a small quantity of fluid samples (Zhu *et al.*, 2020).

h) Track and trace next generation sequencing and meta-barcoding to identify invasive alien species

The COVID-19 pandemic (**Chapter 1**, **Box 1.14**) has demonstrated the effectiveness of real-time genome sequencing on tracking and tracing the virus through movement trajectories of new mutations and strains and therefore spread of an invading organism. This technology approach is being applied now to the management of biological invasions by building global or regional genomics database of a key invasive alien species, both present or considered a potential threat. This allows a) quick identification of any new detections or introductions in terms of origin and likely pathway of spread (Otim *et al.*, 2018; Suarez-Menendez *et al.*, 2020), b) information on global invasion patterns that can help evaluate invasion and impact risks and pick up local rapid evolution or adaptation to new situations (Tay *et al.*, 2022), c) allow real-time monitoring of strategies for and effectiveness of management actions (eradication, containment or widespread management; Yainna *et al.*, 2021).

5.4.2.3 Future technologies

A number of new technologies are being developed and improved for surveillance, detection, monitoring and automated response. These include biosensors and nanotechnology sensors, Clustered Regularly Interspaced Short Palindromic Repeats (CRISPR) diagnostics, multiplexed diagnostic real-time handheld Point-of-Care (PoC) diagnostic platforms and disease mRNA biomarkers. These are covered in more detail in the **Supplementary material 5.4**.

5.4.3 Intervention technologies

When an action needs to be taken a) to manage a pathway risk or consequence, b) once a new invasive alien species has been detected or in order to eradicate, contain or control it, or c) in order to manage a site or ecosystem to eliminate or reduce the impacts of invasive alien species or restore ecosystem function, then this is an intervention. This section covers tools, technologies and approaches available to support interventions.

5.4.3.1 Pathway management – prevention options

Managing biological invasion pathways across borders supports prevention of arrival of new alien species and establishment and the movement of invasive alien species through trade supply chains. The traveller, effects and trade pathways for invasive alien species movement are through air and sea travel, conveyance and transport, postal mail delivery and transport *via* parcels, luggage and container transported traded goods.

Prevention treatments for planes and ships are now widely recognized and regularly applied. Planes are decontaminated before take-off or treated with insecticides prior to arrival to prevent the entry of pathogens and insects including disease vectors such as mosquitos. Shipping is subject to the International Convention for the Control and Management of Ships' Ballast Water and Sediments (BWM Convention, **section 5.5.1**). The two main management options include ballast water exchange in accordance with regulation D-1, which is an interim option, and compliance with the discharge standard in regulation D-2, which is ultimately what all ships eventually have to comply with. The three main methods for ballast water exchange are the "sequential" method (complete ballast replacement with seawater), "flow-through" method and the “dilution” method performed at sea at an agreed distance from the arrival country (Molina & Drake, 2016). Organisms are physically removed by the flow, while the higher salinity level of open seawater can kill any coastal organisms present in ballast tanks (Santagata *et al.*, 2008). The flow-through method pumps flow through seawater into a ballast water tank at a volume sufficient for least 300 percent water replacement (Molina & Drake, 2016), but can lead to lower exchange efficiencies than the sequential method because of mixing between the influent and effluent water (Noble *et al.*, 2016). Most ballast water management systems actually employ a combination of two or more additional treatments (Werschkun *et al.*, 2014). The first of these is mechanical filtration of larger particles (organisms) with self-cleaning filter systems followed by electrochlorination or UV irradiation. Other treatments include ultrasound, cavitation and heating which can also lead to physical destruction of undesirable organisms. The other important prevention approach for shipping is the management of hull biofouling and specific niche areas of the hull. There is, however, currently no international convention regulating biofouling and only national or local regulations exist in a few jurisdictions such as Australia, New Zealand and California in the United States. The International Maritime Organization does have biofouling guidelines (**section 5.5.1**). Context and effectiveness of biofouling management have been reviewed by Arndt *et al.* (2021).

Prevention treatments for biosecurity risks associated with cargo (traded commodities, packaging and containers) are already in widespread use. These include chemical fumigation particularly of wooden pallets and cardboard packaging (still largely using methyl bromide even though regulations under the Montreal protocol have been agreed for some time to phase it out to avoid ozone damage). The IPPC have approved sulfuryl fluoride as a treatment for compliance for ISPM-15 (FAO, 2018a) and additional options include dielectric heating and non-wooden pallets. It is recognized that sulphuryl fluoride is a highly effective greenhouse gas (Papadimitriou *et al.*, 2008). Replacements for methyl bromide are rapidly being sought to support irradiation and cold treatment along supply chains for horticultural commodities. Phosphine is currently in use as an alternative to methyl bromide and is widely used to treat most durable commodities and for disinfestation of stored seed and food grains (Fields & White, 2002). However, there is phosphine resistance in many stored grain beetle pests and this has been documented in Australia, India, Morocco, Brazil, the United States and China (Benhalima *et al.*, 2004; Chaudhry, 1997; Collins *et al.*, 2003; H. Navarro & Navarro, 2016; Nayak *et al.*, 2003; M. A. G. Pimentel *et al.*, 2009). Phosphine resistance in stored grain pests has been increasing in severity and is a threat to the continued use of phosphine as an effective control method (Schlipalius *et al.*, 2015).

Low atmosphere pressure systems (Paul *et al.*, 2020) and microwaves are also used for horticultural commodities but are not generally accepted under multilateral trade agreements (Gamage *et al.*, 2015). When managing the threats of biosecurity contamination *via* trade for agricultural commodity supply chains, bilateral agreements are increasingly considering system-based approaches where more than one approach or treatment are applied in sequence to minimize the risk of pest and pathogen contamination (IPPC, 2017b; van Klinken *et al.*, 2020). See **Supplementary material 5.6** for further details.

5.4.3.2 Species-based management technologies

a) Mechanical and manual control of invasive alien invertebrates and plants

Mechanical and manual management takes many forms and operates at different scales, but in general primarily applies to the management of invasive alien plants (Liebman *et al.*, 2001; Csiszár & Korda, 2015; Hussain *et al.*, 2018). Physical management, like chemical management, if not targeted or sustained is more often ineffective than effective particularly if the seedbanks of invasive plants are not considered or addressed. Mechanical control is occasionally applied to other types of invasive alien species, such as rabbits in Australia where it has proved to be a more cost-effective option than poisoning (Mutze, 1991).

For invasive alien plants, physical management comes in a number of forms including manual pulling, uprooting, mowing, cut and removal, bulldozing, mulching, debarking of trees from the collar region, ploughing and grazing (DiTomaso, 2000; Hussain *et al.*, 2018). Mechanical control, in some form can be an effective strategy as part of integrated management of invasive alien plants or/and ecosystem restoration and some "best practice" can be developed for some targets (e.g., S. King *et al.*, 1996). A key understanding is a preference to limit levels of ecosystem disturbance during the treatment, because more disturbance will assist invasive alien plant reinvasion without additional ecosystem restoration activities (DiTomaso, 2000). Generally human applied physical management approaches for invasive alien plants can be effective locally but are not generally practical or effective at larger scales, particularly when only cutting is used, because targeted species, particularly with large rhizomes, resprout and regenerate (e.g., Mwangi & Swallow, 2005). Moreover, the short-term efficacy and the necessity for periodic implementation for sustained control makes physical methods such as cutting uneconomical. An exception to this is the large scale Working for Water Programme in South Africa (**Box 5.19** in **section 5.5.5**) where the programme, has also offered many additional ecosystem service and social benefits for the local communities supported by government investment (Richardson & van Wilgen, 2004; van Wilgen & Wannenburgh, 2016).

Using grazing animals for invasive alien plant control has a huge literature in the pastoral sector (R. G. Smith *et al.*, 2006; Popay & Field, 1996) and also in rangelands (DiTomaso, 2000; Frost & Launchbaugh, 2003; Sheley *et al.*, 1996), but it has also been used in forests (S. N. Adams, 1975) and other ecosystems (Randall, 1996). Most grazing management occurs through the use of livestock species but, in natural ecosystems, goats are often used with obvious risks of grazing native plants as well or escaping containment, which demands care (Frost & Launchbaugh, 2003; R. G. Smith *et al.*, 2006). The use of grazing animals can also spread invasive alien plants species adapted to attach to animal hides or when invasive alien plant seeds are consumed but may pass into animal faeces (e.g., *Vachellia nilotica* (gum arabic tree); Kriticos *et al.*, 1999). Like other forms of physical management of invasive alien plants, the effectiveness of grazing management generally depends on long-term controlled application as the benefits can be quickly lost when the treatment is halted. This limits control cost-effectiveness, unless part of an integrated management or/and ecosystem restoration programme, where there are multiple other benefits for the local communities (Reyes-García *et al.*, 2019).

b) Pesticide management of invasive alien animals and plants

Chemical pesticides remain a key local tool for invasive alien species management. Herbicides can be sprayed on grasses, herbs and shrubs, applied on cut stumps to prevent sprouting, or injected in tree trunks. For invasive alien plants this is the most widely used control method and a wide array of herbicides are available on the market. Different products are used to control invasive alien animals, especially baits under controlled situations. Social acceptability, risks of non-target impacts and wider environmental risks are increasing societal issues and concerns associated with pesticide use. When addressing these societal issues, it is important to take into account the context of values and policies of individual countries. In order to increase the level of safety and acceptability, it is important that control projects include training and protocols to ensure the use of proper personal protective equipment, good quality materials to prevent leaking, proper disposal of pesticide containers, necessary permits, trained personnel and follow-up routines to increase effectiveness. The use of pesticides is also context-dependent, so local conditions need to be assessed to define limitations. For example, as proximity to water is a strong concern, herbicide spraying regulations require minimum distance from water bodies be maintained. This is equally true for vertebrate pest control programmes. The aerial application of the brodifacoum bait to eradicate *Rattus norvegicus* (brown rat) was conducted on the 267-hectare Ulva Island in New Zealand. This resulted in a residual concentration in several coastal species of fish and shellfish (Masuda *et al.*, 2015). Although the concentrations found were low as was the risk for human exposure to the chemical, this example highlights the importance of being aware of the potential side effects. Similar programmes in Italy appear to have not raised such concerns (Capizzi *et al.*, 2016).

There is a large literature dedicated to applications of chemical use in invasive alien species management beyond the scope of the IPBES assessment of invasive alien species. Most best practice manuals seek to optimize and minimize use as well as choose active ingredients and surfactants that degrade fairly quickly and do not contaminate the soil or water. The use of pesticides in invaded natural ecosystems often generates concern so needs to be highly regulated, particularly close to water bodies to avoid off target impacts (Kolpin *et al.*, 1998). In most countries, use depends on local jurisdictional legislation, registration and approval processes. Approval is needed from regulators to use a particular type of chemical or active ingredient, they also approve how the chemical is to be stored and applied and the targets against which it can be used. This is particularly challenging in the context of the use in natural ecosystems, where across jurisdictions generally few registrations exist and this is a problem for management (Pergl *et al.*, 2020). Whereas pesticides developed for controlling alien fish in rivers are very unselective (e.g., Rotenone used in some countries but banned in others), in China an organophosphate pesticide has been evaluated for its selective efficacy against *Oreochromis niloticus* (Nile tilapia) compared to many non-target native species and based on good results it is being proposed to control this target in the wild where impacts on aquatic ecosystems have been very high (Gu *et al.*, 2018, 2019). In an environmental context, the incorrect (e.g., not complying with regulatory and label restrictions) or non-strategic use of pesticides (how and where they are applied) is probably one of the largest causes of failure to achieve effective local management of invasive alien species in space and time (D. Pimentel *et al.*, 1992; D. Pimentel

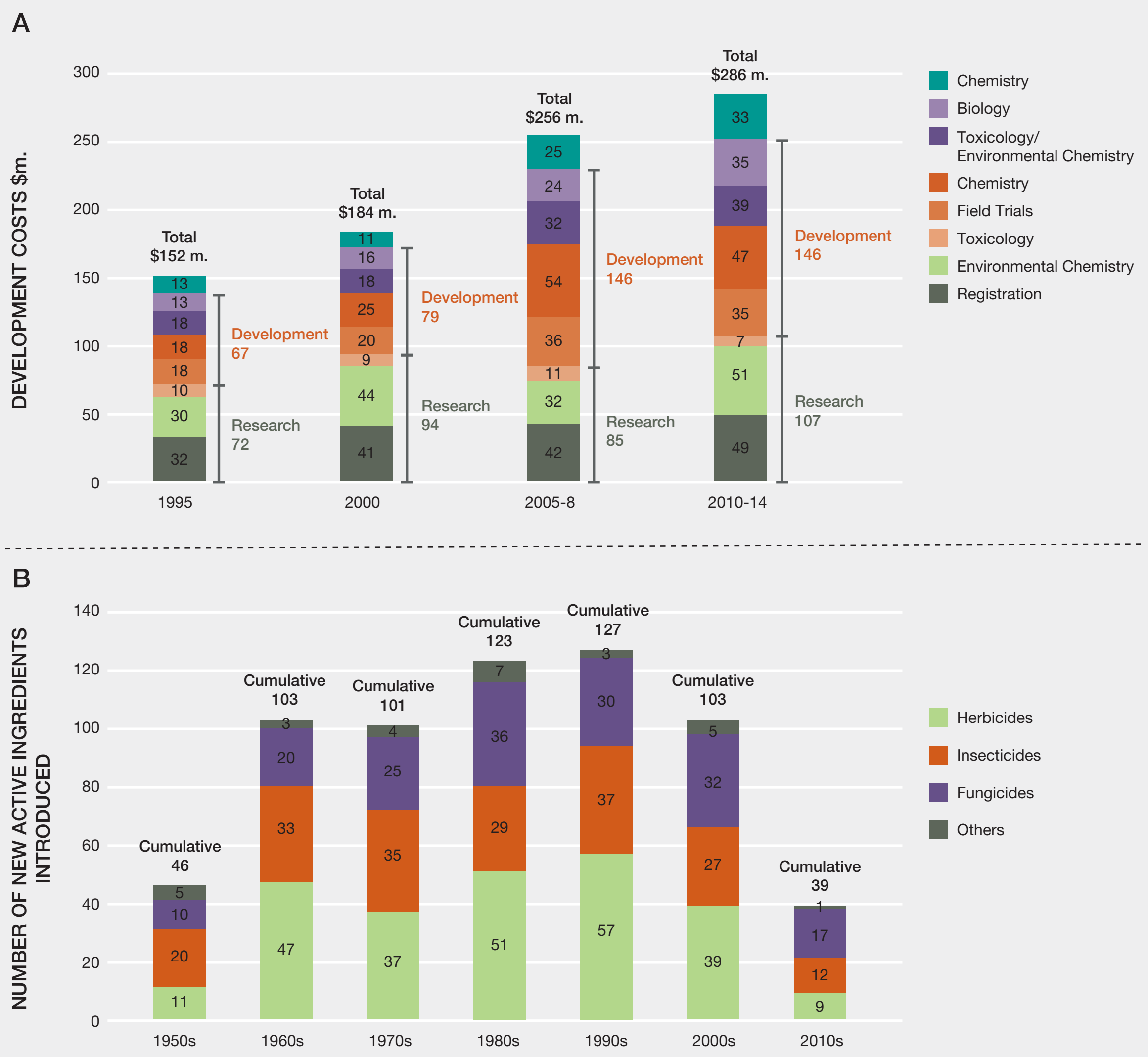


Figure 5 16 **Evolution of new crop protection product.**

Top: discovery and development cost of a new crop protection product (y axis) in 1995, 2000, 2005-8 and 2010-14 (x axis). The overall costs of discovery and development of a new crop protection product increased by 21.1 per cent from $152 million in 1995, to reach $184 million in 2000. From 2000 to 2005-8 period, costs increased by 39.1 per cent to $256 million. From 2005-8 to 2010-14 period, costs increased by 11.7 per cent to $286 million. Source: McDougall (2016), https://croplife.org/wp-content/uploads/2016/04/Cost-of-CP-report-FINAL.pdf, under license CC BY 4.0. Bottom: number of new active ingredients introduced (y axis) per decade (x axis): 1950s to present day. Source: McDougall (2018), https://croplife.org/wp-content/uploads/2018/11/Phillips-McDougall-Evolution-of-the-Crop-Protection-Industry-since-1960-FINAL.pdf, under license CC BY 4.0.

& Andow, 1984) and certainly, results in most human and environmental harm and associated loss of public acceptability associated with their ongoing use. Ongoing use of chemical control needs to be highly regulated and part of effective adaptive management strategies (**section 5.4.3.3**).

Registering new active ingredients has become so lengthy and costly demanding a very large market to be viable and so very few novel pesticides are currently seeking registration (E. D. Booth *et al.*, 2017; Nishimoto, 2019; Phillips McDougall, 2016; **Figure 5.16**). Most new registrations are around new formulations for existing pesticide groups. The main challenges are a) application approaches need to minimize the build-up of pesticide resistance and the global spread of pesticide resistant invasive alien species genotypes (Beckie *et al.*, 2019; Sparks & Nauen, 2015); b) increasing deregistration of key chemical active ingredients (e.g., organophosphates, nicotinoids and glyphosate; WHO, 2010; Sharma *et al.*, 2020) leading to increased illegal off label use (Galt, 2010); c) designing treatment regimes to minimize off target effects (e.g., bitou bush in Australian Coastal heaths –Flower, 2004; Vranjic *et al.*, 2012), d) use in aquatic environments (very few chemicals or adjuvants approved for this; Mesnage & Antoniou, 2018; Grung *et al.*, 2015) and e) an absence of registered herbicides in developing countries can be a major impediment in the control of invasive alien species (Handford *et al.*, 2015). It is critically important that all chemical interventions are undertaken under the regulations for that application and with a strong safety culture for the application staff and the environment (FAO & WHO, 2014).

Nanotech has the potential to reduce pesticide application rates through efficient delivery paving the way for novel applications, devices and systems for delivery of pesticides for invasive alien species management *via* development for agriculture (Manjunatha *et al.*, 2016; Anandhi *et al.*, 2020). Current interest is on three formulation types: polymer-based nano-formulations, inorganic nanoparticles such as silica and titanium dioxide and nano-emulsions. These novel formulations allow the release of active ingredients in a slow and targeted manner, protecting them against degradation and increasing "solubility" of even poorly water-soluble formulations (Manjunatha *et al.*, 2016). Potential risks of nano particles to human and environmental health (the so-called "nanomaterials paradox") has led to delays in application as national nanotech policy and regulatory risk analysis protocols are agreed so the risks are appropriately evaluated (OECD, 2012; Kah, 2015; Agathokleous *et al.*, 2020).

With regard to biopesticides, which come under the same regulatory registration process as chemical pesticides, a review (Glare *et al.*, 2012) stated "Biopesticides based on living microbes and their bioactive compounds have been researched and promoted as replacements for synthetic pesticides for many years. However, lack of efficacy, inconsistent field performance and high cost have generally relegated them to niche products. Recently, technological advances and major changes in the external environment have positively altered the outlook for biopesticides. Significant increases in market penetration have been made, but biopesticides still only make up a small percentage of pest control product". Biopesticides are being applied, for example, against some moths of economic importance (Shao *et al.*, 2018), however they have not been commercially viable against invasive alien plants (Arora, 2003).

c) Robotic technology for targeted management

Robotic technology is increasingly applied to invasive alien species management although this will be challenging in many developing countries without support from international aid programmes. The benefits for management and control activities may be most marked in remote, inaccessible, or broad-scale applications where human actions are costly or dangerous. Robotics can be used to map and characterize invasive alien species (**section 5.4.2.2**) and to deliver a management action (e.g., the application of foliar or granular herbicides; (Carwardine *et al.*, 2016). An autonomous robot can be deployed to operate continuously and may result in significantly less use of chemicals and a reduction of the costs and generally harmful environmental impacts of manned vehicles. Unmanned robotic vehicles are suited to a wide range of aerial, terrestrial and marine applications and can have particular potential for deployment in the management of established pests, weeds and diseases (Jurdak *et al.*, 2015) including marine invasive alien species, but the current technology will be limited for most sessile slow-moving species (D. Smith & Dunbabin, 2007). Robotic platforms can be customized relatively easily to specific tasks and machine learning and artificial intelligence algorithms continue to advance (Devitt *et al.*, 2017). In the field of agriculture, autonomous unmanned ground-based vehicles exist that, in addition to programmed target detection and rapid learning to identify new targets with high accuracy, also include the capability for management decisions and actions. Unmanned ground-based vehicles-based systems are in prototype or in commercial use for many agricultural tasks including pest, weed and disease management, using a range of methods which may be applicable to some invasive alien species management situations. Unmanned Aerial Systems are also being used to deliver pesticides with high levels of precision (Bawden *et al.*, 2017; Lee *et al.*, 2014). There are also robotic technologies currently available for in-water hull cleaning to remove biofouling (**section 5.4.3.1**). Similarly, a robotic automated underwater vehicle developed for environmental monitoring adapted and tested against an undesirable sea urchin on the Great Barrier Reef (Clement *et al.*, 2005; Dayoub *et al.*, 2015) and similar technology is being applied to address vessel in-water biofouling (Scianni & Georgiades, 2019; Tamburri *et al.*, 2020)

The robotic control of unmanned vehicles can be customized to each new application, meaning that each functional system has a high level of invasive alien species target specificity. Systems exist that either recognize the crop and treat all other green material or have learnt to identify specific invasive alien species (Bawden *et al.*, 2017). For example, robotic technologies are starting to be used for weed treatments in crops, pastures and national parks (Olsen *et al.*, 2019; Westwood *et al.*, 2018). Robotic systems for weed management have also been applied against *Vachellia nilotica* (gum arabic tree) in savannas **(Box 5.12)** and alligator weed in rivers (Göktoğan *et al.*, 2010).

Considering system complexity, development costs and sophistication, such technology can only become cost-beneficial and applicable above a certain land value threshold. In the field of marine pest control, the high cost of alternative approaches (including the deployment of divers) may increase the cost-effectiveness. The full costs for the development of an automated underwater vehicle with a robotic injection system for controlling the native crown-of-thorns starfish have not yet been accrued (Dayoub *et al.*, 2015).

Robotics technology can now be bought off-the-shelf. While no full setup for particular invasive alien species applications is readily available, many could be adapted, as demonstrated by several proof-of-concept complexity-testing bespoke systems for invasive alien species management (Ball *et al.*, 2015; Jurdak *et al.*, 2015; Bawden *et al.*, 2017). Technical challenges remain in the field of autonomous on-ground navigation – in particular, using real-time perception and decision-making in harsh environments (Carwardine *et al.*, 2016). Also, requirement of permissions over private land, full control and the "Visual line of sight" in many countries limits the use of unmanned particularly aerial systems. As computing power increases, sophisticated real-time data processing and decision-making using machine learning algorithms will enable a platform to be targeted at particular tasks (Devitt *et al.*, 2017). Current bespoke

Box 5 12 **Case study: Management of *Vachellia nilotica* (gum arabic tree) with robotic technology in Australia.**

In management trials in Australia's Desert Channels region of western Queensland, unmanned ground-based vehicles were deployed for low density infestations of *Vachellia nilotica* **(Figure 5.17)** whereas for poor access infestations, autonomous unmanned aerial systems were deployed for both foliar and granular herbicide applications. Autonomous unmanned ground-based vehicles were used for spraying of low-density *Vachellia nilotica* infestations or configured specifically for spraying over dense stands in open habitats and desert channels. This can increase precision, especially for areas requiring flight beyond visual line-of-sight, reducing both the costs and amount of herbicide, and so the non-target impacts. Still the development costs are extremely high (Carwardine *et al.*, 2016).

Figure 5 17 ***Vachellia nilotica* (gum arabic tree) invading Australia's desert channels region of north-eastern Australia.**

Photo credit: Sahil Ghosh, Adobe Stock – Copyright.

systems are complex, whereas use cases are often quite simple tasks. Any current limitation for adaptable data processing systems is likely to be quickly resolved as the technology continues to mature.

d) Lethal control of invasive alien vertebrate pests

Despite lethal control being the basis of nearly all successful invasive alien vertebrate population suppression (Robertson *et al.*, 2017) and eradication programmes (Holmes *et al.*, 2019) and remaining an important tool for managing their impacts on native vertebrates (see below), ethically, it is increasingly controversial (e.g., van Eeden *et al.*, 2020; **Chapter 1**, **section 1.5.3**). Many countries have animal rights laws and have banned or are banning lethal control options and deregistering key toxicants such as sodium fluoroacetate (1080) and Rotenone. Engaging stakeholders is important for decision-making, particularly assessing the acceptability of lethal control particularly for invasive alien species that cause the loss of threatened and endangered native species and harmful impacts on economic livelihoods of local communities (Deak *et al.*, 2019; Sinclair *et al.*, 2020, **Box 5.13**; **Chapter 4**, **section 4.5**). Often politics controls the decision-making, as in the case for management of alien *Hippopotamus amphibious* (hippopotamus) populations in Colombia. Despite high densities there has been limited political will to cull this charismatic species (Castelblanco-Martínez *et al.*, 2021).

Nonetheless, lethal control is still a conventional approach in countries where large populations of invasive alien vertebrates have massive ecological impacts and cause high native species extinction rates. Lethal control has been the main basis of highly successful vertebrate eradication programs on islands around the world (Holmes *et al.*, 2019; B. A. Jones *et al.*, 2016). On large land masses, lethal control can also be effective at reducing the impact of

Box 5 13 The conflicts of lethal control of invasive alien vertebrates' case study: managing wild horses in the Australian Alps National Park.

Equus caballus (horse) is alien to alpine Australia, but wild horse (local name "Brumbies") populations exist there since grazing properties were first allowed on these Indigenous lands in the mid 1800 **(Figure 5.18)**. Since then, wild horses have been enshrined in classic Australian literature and, even after alpine cattle grazing was banned in the 1990s, horses were left unmanaged. Following this, the Indigenous Peoples and local communities and other communities have been in conflict over whether the horses should be cherished or removed. Culling programmes stopped in 2002 and in 2018 these horses were protected under heritage state legislation. Today numbers have increased seven-fold since 2002 clearly decimating the unique habitats of threatened native alpine species (broad-toothed rat, corroboree frog and she-oak skink) with horse rehoming (the only allowed management strategy) proving insufficient (Driscoll *et al.*, 2019). Choice between native and charismatic alien species is always hard, but the Australian Royal Society for the Prevention of Cruelty to Animals (RSPCA, 2021) has decided to support an aerial culling programme, much to the dismay of the horse lovers.

Figure 5 18 ***Equus caballus*** **(horse) invading the native alpine grasslands of the Snowy Mountains National Park in Australia.**

Photo credit: ms_pics_and_more, Shutterstock – Copyright.

invasive alien vertebrates across short time frames (up to a few years) applied by hunting, poisoning or trapping. It has a long history to support its cost-effectiveness where expertly strategically planned and implemented (Burrows, 2018), but this is rarely the case (Hone, 2007). Where effective, the environmental benefits are well-recognized, even though benefit-cost analyses are rare and are subject to much uncertainty (Newsome *et al.*, 2017). Effective widespread control can be achieved with a good understanding of the local distribution, abundance and population connectedness and planning management at an appropriate spatial scale, together with continual investment (Lurgi *et al.*, 2016). The niche created by removing one invasive alien vertebrate may be quickly filled by immigrants of the same species or by a different species.

Shooting carried out by trained professionals can be a highly-target-specific technique and can minimize the number of animals that need to be culled. In the case of large herbivores or omnivores (including pigs, goats, donkeys, camels, cattle and horses), helicopter-based aerial shooting can achieve rapid population control over large areas. This is less effective where vegetation is dense and not so effective against forest dwelling species such as deer, where ground-based shooting by specialist hunters is more commonly used. Shooting campaigns often involve attaching a tracking device to individual often sterilized animals for social species. Making use of their gregarious nature the tagged animal locates other conspecifics to improve cull efficiency especially at low density. This technique (controversially often termed as the "Judas" approach) has been demonstrated to be highly successful against invasive alien populations of, for instance, *Capra hircus* (goats; K. Campbell & Donlan, 2005), buffalo and feral cattle (More *et al.*, 2015), donkeys (Woolnough *et al.*, 2012), *Camelus* spp. (camels; Edwards *et al.*, 2016) and less effective against pigs (Ramsey *et al.*, 2009). It has also been used against invasive fish (Bajer *et al.*, 2011).

Culling can enhance fertility through earlier reproduction or select for phenotypes that make control more difficult (Newsome *et al.*, 2017). Having the public hunt invasive alien species through the sale of licenses or tags, while potentially cheap, generally leads to perverse trade-offs. Hunters generally reduce their effort as target numbers decline to perpetuate their benefits or there are conflicts of interest when targets are also a food source for the hunter community. Reintroductions into cleared areas are a common occurrence in such situations (Rondeau, 2001). For these reasons, without good collective understanding and planning the benefits of recreational ground-based hunting for invasive alien vertebrates will be largely unquantified and elusive (Bengsen & Sparkes, 2016).

Lethal baiting is a key tool in New Zealand's Predator Free 2050 campaign to remove rodents, possums and stoats (Russell *et al.*, 2015). The combination of toxic bait and delivery system can provide very targeted and species-specific delivery. Available poisons include acute toxicants (e.g., sodium fluoroacetate – or 1080, sodium nitrite or zinc phosphide and para amino-propiophenone (PAPP)), anticoagulants (e.g., pindone and brodifacoum) or fumigants (e.g., carbon monoxide and phosphine; L. McLeod & Saunders, 2013). Zinc phosphide baits are commercially supplied for broad-scale control of mouse plagues and remain an effective way of treating large areas in a relatively short period of time. Electrofishing and piscicide (an odourless, colourless, crystalline isoflavone) are the dominant means used to control invasive alien fish. Rotenone is being de-registered in some jurisdictions because of non-target impacts (McLeod & Saunders, 2013). Widespread and high-density use of these toxicants remains controversial for a number of reasons but is generally highly regulated by jurisdictions to address concerns. Lethal control in a conservation context has been very effective for rodent and rabbit eradication programmes on islands (S. Gregory *et al.*, 2014; Howald *et al.*, 2007; Russell *et al.*, 2015). Island use has continued to be refined, with an increase in the size of island from which rodents can feasibly be eradicated (Howald *et al.*, 2007; B. A. Jones *et al.*, 2016). In most other settings the likelihood of total eradication of an invasive alien species is low. There are reviews of eradication success (Mill *et al.*, 2020; H. P. Jones *et al.*, 2016) and the European Union has guidelines for the eradication of invasive alien vertebrates (Genovesi, 2001). Prolonged use of a single toxicant leads to development of resistance (Twigg *et al.*, 2002). Registration of all toxicants is required for each jurisdiction covering environmental safety, efficacy and relative humaneness with required label registration for use against each specific target. Nonetheless, development of new toxicants for vertebrate pest control continues in the United States, Australia, New Zealand and Europe along with baits and delivery methods specific to particular target species (Begley *et al.*, 2021; T. A. Campbell *et al.*, 2013; Eason *et al.*, 2017; Robertson *et al.*, 2017). In the future, the use of existing self-resetting traps (Carter *et al.*, 2016; Stanley, 2004), toxicant delivery systems, advanced lures and new toxicants with lower residues could be improved. Ongoing use of the same bait-based toxicants causes bait shyness (neophobia) in surviving populations (Garvey *et al.*, 2020). Effectiveness of baiting and trapping varies greatly with the target alien invasive vertebrate species. Cats, rats and to a lesser extent wild dogs are notoriously bait and trap shy and need to be tricked into bait taking (Garvey *et al.*, 2020).

Camera trapping can be used to improve trapping outcomes (Fleming *et al.*, 2014; Meek *et al.*, 2015) and is advancing rapidly with increasingly complex algorithms allowing individual animal recognition either from patterns or facial features. Grooming traps (or sentinel automated spray devices) use shape recognition technology designed to recognize species (e.g., cats) and spray a lethal amount of a toxicant only on the target animal which is then ingested by grooming. Ejector technologies include direct delivery of a lethal toxicant into the mouth of a predator by combining a spring-loaded mechanism with a carnivore lure, which is more likely to be activated by foxes and wild dogs, thereby increasing target specificity (Fleming *et al.*, 2006). Lethal control methods are also best used as part of an adaptive and integrated management approach that aims to reduce the overall quantity of toxicants entering the environment.

e) Fertility control for invasive alien vertebrates

Fertility control tools aim to decrease or stabilize a pest animal population by reducing or halting reproduction. Modelling suggests that targeting females has the greatest chance of effective control at the population level (Caughley *et al.*, 1992). Successful fertility control reduces target population sizes unless flow-on effects (e.g., increased survival of adults and juveniles) exceed the effects of reduced fertility. Fertility is controlled through a) the disruption of key reproductive hormones, b) the use of chemicals to deplete follicles at various stages of development, or c) immune contraception approaches where an immune response is elicited to key reproductive antigens that subsequently interferes with fertilization and/or successful embryo implantation. Effective hormonal or immune contraception methods are available; however, they require the capture or restraint/sedation of the animal to apply the treatment either *via* injection or implant followed by release, or remote delivery *via* darts, which is generally at high cost. With growing public concern about lethal control methods, public acceptability for fertility control approaches is usually high, for high-profile, iconic species.

Fertility control has been applied in the United Kingdom, United States, New Zealand and Australia in rodents, goats, horses, deer, kangaroos and canines (Asa & Moresco, 2019; Cowan *et al.*, 2020) but generally still at very localized scales. There remain very few cases where fertility control has been successfully used to control free-living populations or to achieve eradications. Nonetheless, this approach potentially has broad range of application in vertebrates, limited primarily by the lack of effective bait delivery systems useful in the wild (T. A. Campbell *et al.*, 2011). Other challenges include optimum dose levels, stakeholder opposition (e.g., hunters) and sex specific welfare implications. Applications requiring oral bait delivery or self-disseminating fertility control agents still await development. Applications to-date are principally in containment and control, although fertility control can also aid in eradication of small populations (Hobbs *et al.*, 2000). Surgical sterilization has been used in the management of grey squirrels (Scapin *et al.*, 2019) and bullfrogs (Descamps & De Vocht, 2017) in Europe. For registration, commercially available fertility control vaccines must undertake delivery and effectiveness trials, which generally require a minimum three to five years. See **Supplementary material 5.6** for further details.

f) Classical biological control of invasive alien plants and invertebrates

Classical biological control has an over 100- year history and has been accepted as a long term and effective management tool for invasive alien species by both the IPPC (IPPC, 2017a) and the CBD (ISSG, 2018), particularly for invasive alien plants and invertebrates (ISSG, 2018; Julien & White, 1997; Waterhouse & Sands, 2001; Heimpel & Mills, 2017) in both agricultural and environmental settings (Van Driesche *et al.*, 2010). Classical biological control programmes aim to release host-specific natural enemies of an invasive alien species target, generally from the native range and suited to the recipient environment of the target (Briese, 2000). Classical biological control has not been developed for marine environments, because of safety concerns partly because they are globally contiguous and in general less understood than terrestrial or freshwater ecosystems (Lafferty & Kuris, 1996; Thresher *et al.*, 2000; Secord, 2003). While some consider it remains an option worth considering (Bax *et al.*, 2003), others consider the use of alien biocontrol agents too risky in marine systems (Atalah *et al.*, 2015).

The application of biological control dates back to the uncontrolled use of generalist vertebrate predators as biocontrol agents from the 1700s to early 1900s prior to import restrictions on alien species or regulations built on robust risk assessment (Huffaker & Messenger, 1976; Moran *et al.*, 2013; Waterhouse & Sands, 2001). This includes the release of cats to control rodents, mongoose to control snakes on islands and toads to control sugar cane pests. This gave biological control a bad name because many of these agents became pests in their own right (**Chapter 3, section 3.3.5.2**). These are not examples of classical biological control as these "biocontrol agents" were not selected based on specificity (some being generalist fish; e.g., Fenichel *et al.*, 2010; Ip *et al.*, 2014) or *via* a risk-assessment-based regulatory process and often released against native pests. Classical biological control now has mature regulatory processes for the identification of agents, and for the importation, assessment and release of agents in recipient countries (A. W. Sheppard *et al.*, 2003; Day & Witt, 2019; Barratt *et al.*, 2021) built on internationally agreed best practice principles and guidelines (IPPC ISPM-3; Sheppard *et al.*, 2003; M. Day & Witt, 2019; Barratt *et al.*, 2021). The risk of non-target impacts has been given considerable attention with some considering them too high (e.g., Simberloff & Stiling, 1996), a lead critic has since recognized that the benefits merit careful ongoing use (Van Driesche *et al.*, 2010). The specific and broad international benefits and undesirable non-target impacts of historical biological control programmes are well documented and the benefits far outweigh the risks in the vast majority of programmes (CBD, 2018). Under the CBD, biological control activities now need to take into account access and benefit sharing regulated under the Nagoya protocol, however as a discipline biological control has embraced this process and continues to operate (P. G. Mason *et al.*, 2021).

Successful classical biocontrol agents have included the following:

- Biotrophic fungi for plant targets (particularly rusts, e.g., *Puccinia* spp.) and arthropods (*Beauveria* or *Metarhizium* strains) (Hershenhorn *et al.*, 2016; Morin, 2020; Morin *et al.*, 2006)
- Invertebrate predators or parasites of invertebrate alien species such as parasitoid wasps and flies, entomopathogenic nematodes (Hajek *et al.*, 2007; P. G. Mason, 2021; Waterhouse & Sands, 2001)
- Viruses to control certain invertebrates, such as *Oryctes rhinoceros* (coconut rhinoceros beetle) across the Pacific (Paudel *et al.*, 2021).
- Herbivorous invertebrates from a broad range of groups for invasive alien plants (Julien *et al.*, 2012; McFadyen, 1998; Winston *et al.*, 2014).

Classical biological control programmes can only be initiated against a specific invasive alien plant or invertebrate if there is broad agreement across different stakeholder communities (scientists, conservation organizations, other land managers, policy makers and the general public) and Indigenous Peoples and local communities of the harmful nature of the target. This process needs to consider social, economic and environmental impacts and any conflicts of interest (**section 5.2**). Where the target has spread across multiple jurisdictions with contiguous borders, affected jurisdictions also need to agree about the target and the regulatory processes under which the biological control programme will operate. A precautionary approach may be adopted because classical biological control is invariably of high risk, biocontrol agent releases generally being without controls and self-perpetuating. As classical biological control programmes are not always successful, the public need to understand this. The regulatory process prior to any release decision being made needs to include effective risk communication between all stakeholder communities. The use of structured decision-making processes, supported by rigorous cost-benefit and risk analyses, not only provide a sound broadly accepted rationale for investment in what is generally a long-term activity but also to contribute to the credibility and success of this approach (S. Liu *et al.*, 2011).

Classical biological control is generally not used for eradication, which is a very rare outcome even as part of integrated control (e.g., Morin *et al.*, 2009). Only recently has classical biological control been developed pre-emptively for invasive alien species before establishing in a country and only when eradication is very unlikely following establishment (Charles *et al.*, 2019).

Classical biological control programmes generally have four phases: i) agent selection, ii) agent risk assessment, iii) agent release and iv) post release evaluation (CBD, 2018; Heimpel & Mills, 2017). Each phase may require two to five years.

Agent selection is sourcing a potentially suitable biocontrol agent in the native range of the invasive alien species target. Genetics, such as barcoding, has assisted this reducing the time required to identify and classify candidate classical biological control agents.

Risk assessments for biological control agents evaluate host-specificity of the agent using internationally accepted protocols (Bigler *et al.*, 2006; ISSG, 2018), either in the native range or a post entry quarantine facility in the country of introduction that considers:

- direct impact of the biocontrol agent on non-target species;
- potential for indirect impacts of the biocontrol agent, including effects on organisms that depend on the target pest and non-target species and competition with resident biocontrol agents and other natural enemies (not all practicing jurisdictions require this);
- possible direct or indirect impact on threatened and endangered species, ecosystems, agriculture and forestry, in the country of introduction;
- impact of the biocontrol agent on humans (health, social and cultural), and impact of the biocontrol agent on the physical environment (e.g., water, soil and air).

Risk assessment is generally less onerous for secondary jurisdictions if a biological control agent has been widely tested and established safely in numerous countries.

Agent release can only happen following application for and approval of a release permit by an independent regulatory body based on the risk assessment. Release submissions generally require public comment, scientific peer review and consultation with neighbouring countries before making a decision. The decision assumes that the agent will have unlimited uncontained spread across the target population. Releases may require modification of the recipient environment (e.g., initial cages or nutrient levels) to improve the control agent's establishment and spread.

Post release evaluation is critically important to measure both positive and negative impacts of the biological control agent. Such impacts may appear within two years, but full effectiveness may also take a further 10 or more than 20 years. The best measure of success likelihood is if agents have already been successful in other jurisdictions against the same target. Generally, more than 50 per cent of classical biological control programmes deliver some level of success and benefit cost ratios of total jurisdictional investments in this approach are generally above 20:1 (ISSG, 2018).

Advantages of classical biological control include its relative cost-effectiveness and broad-scale, long-term, non-chemical and target-specific application. The initial implementation costs are generally high compared to, for example, manual adaptive management approaches, but not compared to new pesticide registration and control when it occurs is generally widespread and enduring.

International collaboration is critically important in classical biological control programmes for the following reasons:

- To respect the Nagoya Protocol, as biocontrol agents are generally sourced from the native ranges (in other countries) of invasive alien plants and invertebrates;
- To avoid released biocontrol agents spreading across international borders;
- To share experience and approaches in classical biological control between experienced and inexperienced countries;
- To save considerable time and costs in controlling savings based on sharing of research, control agents, risk assessments and funding for control programmes against shared invasive alien species.

Classical biological control for invasive alien species management is not a profit-making activity and so, outside of agriculture, is usually funded by public or not-for-profit agencies with responsibilities in environmental management. Countries in North America have also worked together under the North American Plant Protection Organization (NAPPO) to take a regional (continental) approach to collectively manage biological control release activities that affect multiple jurisdictions.

The public acceptance of classical biological control globally still remains mixed, despite a high success rate in countries such as Australia, New Zealand, South Africa, Canada and the United States and an extensive history across many countries (Winston *et al.*, 2014; Cock *et al.*, 2016; P. G. Mason, 2021). Some countries have high public and regulatory perceptions of risk around the use of pathogens as biological control agents (for example, United States) even though forty years of evidence elsewhere suggests otherwise (ISSG, 2018). Some classical biological control agents have been released even in the recent past without the application of a rigorous risk based precautionary approach and have led to significant non-target impacts that were avoidable (for example, *Harmonia axyridis* (harlequin ladybird) in Europe; H. E. Roy *et al.*, 2016). Persistent concerns about non-target impacts continue to be addressed through learnings from past practices (Follett & Duan, 2000; Louda *et al.*, 2003) and through rigorous risk assessment (ISSG, 2018; P. G. Mason, 2021).

g) Sterile insect technique and other relevant invasive alien invertebrate augmentative approaches

Sterile Insect Technique is based on mass releases of irradiated infertile males and is a mature technology based on conventional approaches that have proven to be effective for 65 years (Dunn & Follett, 2017). The infertile males compete with wild males to breed with wild females, and this leads to a reduction in offspring and a decline in population numbers. Eventual local pest population extinction is possible. Sterile insect technique is primarily used for agricultural pest management but also has a history of success for managing invasive alien disease vectors (FAO & AEG, 2016). The approach was first used in the 1940s to control *Cochliomyia hominivorax* (New World screwworm) and subsequently *Glossina* spp. (tsetse fly), *Pectinophora gossypiella* (pink bollworm), *Cydia pomonella* (codling moth) and *Delia antiqua* (onion fly), with recent applications against mosquitos (e.g., *Aedes aegypti* (yellow fever mosquito), *Aedes albopictus* (Asian tiger mosquito)) as vectors of arbovirus diseases (e.g., Dengue fever) (Dyck *et al.*, 2005; Poncio *et al.*, 2019). The technique has also been applied to crayfish (Aquiloni *et al.*, 2009). The technique requires the production and release of enough sterile males to achieve at least 50 per cent of all matings by wild females and to compensate for loss of fitness caused by the irradiation treatment (Helinski & Knols, 2008; Holbrook & Fujimoto, 1970; Mayer *et al.*, 1998) and is also considered more effective if females are not released (Dyck *et al.*, 2005; A. S. Robinson, 2002). With the advent of modern genetics of fertility novel approaches could broaden out Sterile Insect Technique to a range of other invasive alien invertebrate targets (Choo *et al.*, 2018). Sterile insect technique is frequently used as part of integrated pest management strategy, in combination with insecticides and baiting strategies. See **Supplementary material 5.7** for further details.

h) Viral biological control of invasive alien vertebrates

Viral biological control is a special case of classical biological control where the classical biological control agent is a taxon-specific virus (critical), and the target invasive alien species is (in general) an invasive alien vertebrate. Viral biological control is predicated on the discovery of a suitable viral agent and is generally most effective when the invasive vertebrate target is an animal population naïve to the pathogen. Such pathogens also require high virulence, transmissibility and relative humaneness in the method and speed of kill (when compared to other approved control methods). Over time resistance builds up to an equilibrium of lower viral pathogenicity or virulence as individual and "herd" (population-level) immunity develops. Therefore, viral biological control programmes need a long-term strategy to find new more virulent viral strains for re-release to resuppress target population (Cox *et al.*, 2013; McColl *et al.*, 2016). As with classical biological control, viral-based approaches rarely eradicate a widely established target population. The best results will be obtained when viral biological control is one component of an integrated pest-management strategy that includes other (conventional or novel) approaches. Viral biological control depends on strong public support and a bespoke national regulatory system to vet the non-target and any other environmental risks or international concerns (Cooke & Fenner, 2002).

This approach has only ever been applied for alien vertebrates in Australia and then carried to New Zealand through the release and natural dissemination of the Myxoma virus (MYXv) in the 1950s and Rabbit haemorrhagic disease virus (RHDv) in the 1990s for the viral biological control of *Oryctolagus cuniculus* (rabbits; Cooke *et al.*, 2013; Saunders *et al.*, 2010). That Australia and New Zealand are islands reduces the risk of the pathogen spreading back to the native range of European rabbits, although the viruses released in Australia were already present in the native range from where they were largely sourced. This approach is not appropriate where contiguous land masses or water bodies can allow epidemic spread between targeted invasive and native populations. A contemporary example is the ongoing evaluation of cyprinid herpesvirus 3 (Cy-HV3, carp virus) as a viral biological control agent for European carp in Australian waterways. Despite extensive searches, a species specific lethal natural pathogen of *Rhinella marina* (cane toad) has not yet been identified (Shanmuganathan *et al.*, 2010), demonstrating a key limitation in this approach. Similarly viral biological control of feral cats has been used to aid the eradication of cats from sub-Antarctic Marion Island (Bester *et al.*, 2002), but has little practical use elsewhere as this virus is endemic in most cat populations globally so resistance will be widespread and no others are currently considered suitable (Tracey *et al.*, 2015).

Public acceptance of releasing viruses to control invasive alien species may differ depending on differing socio-political perspectives. Lethal biological control methods for sentient lifeforms such as vertebrate pests continue to raise animal welfare concerns particularly in targets also kept as pets or farm animals. Values are changing. From a humanness perspective Myxoma virus (MYXv) would unlikely be approved for release in Australia today, and RHDv was considered more humane than other approved rabbit control methods due to its extremely fast disease progression (Sharp & Saunders, 2011). Continued use of RHDv nonetheless still encounters resistance, but mainly from outside Australasia. In Australia the approach is supported by the local Royal Society for the Prevention of Cruelty to Animals (RSPCA, 2019). Many animal diseases (e.g., Cy-HV3, carp virus) are notifiable under the WOAH presenting another challenge, although the WOAH also assesses levels of specificity for all notifiable pathogens. Risks can be managed if animal and animal product movement can be controlled and developing a vaccine to protect non-target individuals (i.e., pets) may be prudent (Schirrmeier *et al.*, 1999). See **Supplementary materials 5.6** for further details.

i) RNA Interference

RNA interference (RNAi) describes the process whereby a mirror image double-stranded RNA (dsRNA) molecule of the target gene is created, applied, and once it has entered a cell, will silence or modify target gene expression without genetic modification. RNAi is a widely used molecular technique for selectively inactivating genes without the need to create a special strain or modify the genome of the target organism **(Table 5.5)**. Delivering dsRNA molecules to virtually all eukaryotes induces the RNAi silencing process, whereby the dsRNA molecules mediate a highly-specific destruction of any messenger RNA with a complementary sequence, resulting in suppression of a targeted gene's expression (D. H. Kim & Rossi, 2008). Due to RNAi's sequence-specificity, dsRNA molecules can be designed to be highly species-specific (Baum *et al.*, 2007; Pan *et al.*, 2016; Whyard *et al.*, 2015).

The RNAi approach has been proposed as a novel means by which to control plant pathogens, including viruses (Tenllado & Díaz-Ruíz, 2001; Mitter *et al.*, 2017) and the fungus *Fusarium* (Koch *et al.*, 2017; Machado *et al.*, 2018; Weiberg *et al.*, 2013), invasive alien plants such as Phragmites (reed; Hazelton *et al.*, 2014) and invasive alien ants (Gruber *et al.*, 2017), moths (e.g., *Helicoverpa armigera* (cotton bollworm); Z. X. Lim *et al.*, 2016) and *Aedes aegypti* (yellow fever mosquito; Whyard *et al.*, 2015). RNAi is also applicable to animal, zoonotic and human diseases. Linke *et al.* (2016) used bacterial vectors that targeted avian mucosal epithelial cells to deliver RNAi against two avian influenza genes. In this application, RNAi is injected into the tail of wild-caught female prawns. The offspring of these females can then be introduced into farm ponds with a reduced risk of introducing the diseases circulating within wild prawns – including the devastating disease, white spot. Advantages and disadvantages of RNAi are given in **Table 5.5**.

Mutations in the RNAi machinery of the target organism might compromise RNAi effectiveness (Khajuria *et al.*, 2018).

Table 5 5 **The advantages (left column) and disadvantages (right column) of RNA interference in the context of controlling invasive alien species.**

Summarized from information in Vogel *et al.* (2019)

Advantages	Disadvantages
Exogenous RNA interference does not use a genetically modified technology (free of genetically modified regulatory requirements in most jurisdictions)	Endogenous RNA interference use required genetically modified targets (Subject to genetically modified regulations); exogenous RNA interference may require regulatory review in some jurisdictions
Highly- specific and yet adaptable to many species	Limited to one target invasive alien species for each application
Wide range of potential target genes could be targeted	Requires an annotated genome of target for gene selection
Non-chemical (biological) and hence negligible impact on the environment	Poorly stable in the environment so requires an effective encapsulation and delivery mechanism
Likely social and has market acceptability	Likely to be expensive until biotech develops low-cost production systems
Target resistance development can be quickly countered through realigning RNA interference sequence to the resistance gene	Sequence homology in other species
Can improve, facilitate or supplement existing strategies when used in the context of an integrated pest management strategy	Lack of uptake by some species

Should a target evolve resistance, the dsRNA molecules can be quickly redesigned to circumvent the resistance gene mutations. In plant pathogens such as viruses, where any treatment is difficult, or for fungal pathogens where increasingly virulent strains emerge, or fungicide resistance is developing, RNAi provides strong new potential opportunities. The key challenge in the development of an RNAi application lies in formulating the product to deliver the dsRNA effectively to the target animals and the relevant target cells.

The exogenous use of this approach is generally considered safe and does not require regulation in some countries. The only material concerns seem to be persistence of the formulated delivery systems, but new stable formulations are being developed. If nanotechnology is used, then separate concerns may arise from this. It is anticipated that the technology will gain reasonable community acceptance. As this approach is still relatively new with technical challenges still to be addressed there remain very few cases where RNAi as a topical application has provided effective invasive alien species management (Das & Sherif, 2020). The method could become the next generation of pesticides or may improve the cost-effectiveness by replacing irradiation for the sterile insect technique (**section 5.4.4.2.g**). See **Supplementary material 5.6** for further information.

j) Genetic-control approaches (including gene-drive)

The objective of genetic-control approaches is to reduce the fitness or success of an invasive alien species in its invaded environment. The aim is generally to force the population towards one sex (generally male biased) which, if complete, will lead to extinction (Teem *et al.*, 2020). Research into these methods, particularly for invasive alien animals, has made significant advances towards application (Bax & Thresher, 2009; Gierus *et al.*, 2022; Thresher, van de Kamp, *et al.*, 2014). There are two general approaches to genetic control: (a) exploiting natural genetic variants that lead to sex bias in progeny, such as the Trojan Y-Chromosome strategy (Gutierrez & Teem, 2006); or (b) genetic modifications that lead to sex biases or induce population fitness reductions in other ways. A few are being explored (e.g., *Limnoperna fortunei* (golden mussel); Rebelo *et al.*, 2018), but no off-the-shelf genetic-control tools are currently available. Genetic control could be applicable to most sexually reproducing invasive alien species and, for environmental applications, could address currently uncontrollable widespread established invasive alien species in contained settings (e.g., invasive alien fish in closed river systems or invasive rodents on islands).

Genetic-control approaches have a number of significant advantages:

- Locked in and spread only within sexual reproducing populations i.e., strict species specificity (except possibly in some fungi which exhibit asexual gene transfer).
- Dissemination of control is mediated by the invasive alien species itself.
- No environmental residues where successful eradication is the outcome. Genetic modification is of only a few genes which are naturally broken down on death.

- Potentially effective over large geographic areas (depending on gene flow in pest population).

- Humaneness – these technologies are not lethal if reducing reproduction is targeted (Teem *et al.*, 2020), though strongly sex-biased populations may result in behavioural stress.

The Trojan Y-chromosome strategy in fish (currently under development for carp and tilapia) naturally alters the sex determination chromosomes XX♀ and XY♂ to create sex-reversed super-males (YY♀) which, if continuously released into invasive alien species populations in the field, generate only male or super-male progeny through standard Mendelian inheritance (traits in 50 per cent of progeny) (Teem *et al.*, 2020). The resulting male-biased population could ultimately collapse leading to extinction. Broader application of this approach to other target species will depend on whether they have appropriate sex determination systems.

Synthetic genetic modification allows increased opportunities for sex manipulation. An approach for "daughterless" carp for example has made it to the proof-of-concept stage (Thresher, van de Kamp, *et al.*, 2014). Here sex-biasing gene constructs are implanted in the genome of candidate fish. Their fertile offspring, if released and make up a high enough proportion of the population should, through natural inheritance (a reproductive event where a wild-type passes the gene construct on to 50 percent of offspring), drive populations male-biased. This requires large single or multiple releases of these genetically modified invasive alien species genotypes. Large initial invasive alien species population size increases the cost of application (numbers of modified individuals that need to be bred up and released) and time to control. An expert assessment of genetic options for the management of sea lampreys in the United States prioritized such a Mendelian "sex-ratio drive" approach (Thresher *et al.*, 2019). In theory, this approach is applicable to other invasive alien species (Thresher, van de Kamp, *et al.*, 2014). If this type of control fails (or succeeds) the associated genetically modified organisms will be bred out of the population.

A meiotic "gene-drive" mechanism is one in which inheritance of such genetically modified deleterious gene constructs would be much higher than 50 per cent. This could deliver a step-change in population suppression or eradication rates. Natural gene-drive mechanisms exist. The t-allele is a natural, lethal when homozygous, mutant in mice that is inherited by greater than 50 per cent of progeny of heterozygous male carriers. Genetically linking the t-allele to the male Y sex chromosome (called T-Sry) could also mean a far greater proportion of progeny are male (Kanavy & Serr, 2017). A synthetic T-Sry approach for mice is now at the proof of concept stage (Gierus *et al.*, 2022). There are several other naturally occurring selfish genetic elements in wild populations of most organism types (so called natural gene-drives) that could potentially be modified into self-sustaining meiotic gene-drive systems without the need for synthetic genetic modification (Ågren & Clark, 2018; Wang *et al.*, 2014).

Synthetic genetically modified gene-drive mechanisms have now been demonstrated in mosquitoes (Adolfi *et al.*, 2020) and rodents (Gierus *et al.*, 2022). New CRISPR-based gene-editing tools have provided a step-change for this technology development. Synthetic gene-drive systems could drive any potentially deleterious gene into the population. The highest precision genetic engineering tool so far is a new class of "base editors" (programmable protein machines) that can individually replace all four nucleotides of DNA selectively and efficiently, without the need for double-stranded DNA breaks (Gaudelli *et al.*, 2017). These have the potential to make single nucleotide alterations; the smallest and most precise way to make deleterious modifications to genes for invasive alien species control. This may bring broader applications beyond sex-biasing, for example, altered disease resilience or susceptibility to otherwise innocuous chemical agent etc (Legros *et al.*, 2021). Once constructed, the modified organism needs to be assessed for efficacy and the heritability of the deleterious gene into the invasive population prior to regulatory approval. Effectiveness will be slower for target species that have lower reproductive rates, although releasing more modified individuals may speed up time to effectiveness.

Addressing public acceptability for genetic control tools in their various forms has become an independent research focus (Kirk *et al.*, 2019; MacDonald *et al.*, 2020; Mankad *et al.*, 2022; Simon *et al.*, 2018). To progress this there are several open online networks and forums, open and transparent research principles and codes of ethics that the key research agencies have signed up for both the research and field trials. Risk analysis and addressing public concerns of such approaches is therefore critical for seeking and obtaining approvals and supporting prior and informed consent with Indigenous Peoples and local communities in areas where this technology is being considered for deployment (Taitingfong, 2019). As with all invasive alien species control programmes, ecological and genetic modelling studies are also critically important for pre-evaluating effectiveness and likely impacts for any given invasive alien species (e.g., Birand *et al.*, 2022; Thresher, Hayes, *et al.*, 2014; **section 5.6.3.2**). Regulatory acceptance and approval are also mandatory so as to ensure researchers work closely with regulators from the start to ensure common understanding of the risks and the concerns.

Gene-drives that could spread "uncontrolled" within a species are highly unlikely to be acceptable. There is always a risk of gene transfer into desirable native populations of the same species. Risk management could be through developing and including a genetic mechanism to stop the unlimited spread of synthetic gene-drive carrying individuals. This is a focus of current research and a process of risk analysis has been developed to assess this in detail (K. R. Hayes, Hosack, Ickowicz, *et al.*, 2018). As no successful gene-drive system has been developed and applied, it is too early to consider that uncontrolled gene-drive systems are the only option (Esvelt & Gemmell, 2017). If gene-drives fail to eradicate the target or only suppress target populations. synthetic gene-drive carrying individuals could also theoretically persist in the environment (Champer *et al.*, 2021). Persistence in any form may not be acceptable (Legros *et al.*, 2021).

The United States National Academy of Science Engineering and Medicine has released its landmark discussion paper, Gene-Drives on the Horizon (National Academies of Sciences, Engineering, and Medicine, 2016). A shorter discussion paper was also released by the Australian Academy of Science and Technology, titled Synthetic Gene-Drives in Australia (Australian Academy of Science, 2017). Both reports discuss the practicalities and risks of the science and its application and make recommendations in relation to physical containment. The IUCN has also recently released a report entitled "Genetic frontiers for conservation: an assessment of synthetic biology and biodiversity conservation" reviewing the risks and potential of these technologies (Redford *et al.*, 2019), as has the National Invasive Species Council in the United States (ISAC, 2017). This report includes a number of case studies and chapters on governance. Some more recent relevant reports include: a) Synthetic gene drive: between continuity and novelty (Simon *et al.*, 2018), b) Gene Drive Organisms: Implications for the Environment and Nature Conservation (Dolezel, Simon, *et al.*, 2020) and c) Beyond limits – the pitfalls of global gene drives for environmental risk assessment in the European Union (Dolezel, Lüthi, *et al.*, 2020). See **Supplementary material 5.6** for further information.

5.4.3.3 Site-based management approaches

Site-based management and ecosystem-based management strategies as discussed in **sections 5.1**, **5.2** and **5.3**, aim to suppress the long-term impacts of invasive alien species on biodiversity and ecological assets at that location or in that ecosystem. These approaches tend to integrate invasive alien species suppression and site/ecosystem restoration. This section briefly covers integrated invasive alien species management and the incorporation of restoration science mainly in terrestrial ecosystems. Restoration in marine environments is currently considered to be largely ineffective once invasive alien species are established and spread.

a) Adaptive integrated management strategies

Integrated management of invasive alien species is the equivalent of integrated pest and weed (Hatcher & Melander, 2003) management strategies in an agricultural context. Integrated strategies are where more than one approach is used in combination either in sequence or parallel. This could be a combination of approaches (e.g., chemical and biological) to manage one or more invasive alien species at a given location, or it could mean the integration of invasive alien species management with site/ecosystem restoration or both. What is very important in integrated strategies as for sites or ecosystems is that they are very context dependent. As such they need to be treated as an experiment from the start and developed and extended using an adaptive management approach. Fire is also often used as part of integrated invasive alien plant management in grasslands, savannas and rangelands, but needs to be used with caution (L. Provencher *et al.*, 2007; Weidlich *et al.*, 2020). The African invasive alien plant *Vachellia nilotica* (gum arabic tree) has become a major savanna invasive alien species in Asia and Australia. In Indonesia a management programme has been supported by the United Nations Global Environment Fund in the Baluran National Park in east Java since 2016 (Zahra *et al.*, 2020). Multiple management options of physical action (4 options), fire, biochar, biological (competitors and antagonists) and social (education and adaptation) are being integrated through an adaptive management approach in this ecosystem **(Figure 5.19)**.

While the number of invasive alien species exceeds the capacity for management, not all species pose the same risk to the nature, nature's contributions to people and good quality of life (**Chapter 4**, **sections 4.3**, **4.4**, **4.5**, **4.6**). Thus, it becomes critical to evaluate the feasibility of applying different management strategies, taking into consideration the cost and benefits of each. Unsuccessful programmes do not encourage public and stakeholder support for future actions (Zimmerman *et al.*, 2011), so management decision-making processes (**section 5.2**) bear the responsibility of first assessing the likelihood of success of any management action **(Figure 5.1)**. Management programmes are dynamic in time and space and operate at different temporal scales (Kueffer *et al.*, 2013) as they must change with the invasion status of the target species (e.g., a species' distribution, position on the invasion curve, abundance or impact), and as scientific knowledge and societal perceptions change (e.g., Davies *et al.*, 2020). Management is implemented within unpredictable, complex socioecological systems (Pyšek *et al.*, 2020; R. T. Shackleton, Larson, *et al.*, 2019). This adaptive management approach is fundamental to all effective natural resource management and has two components: (1) to learn and adapt and (2) to do so purposefully with relevant partners (Latombe *et al.*, 2019; Roux *et al.*, 2011). The core principle of adaptive

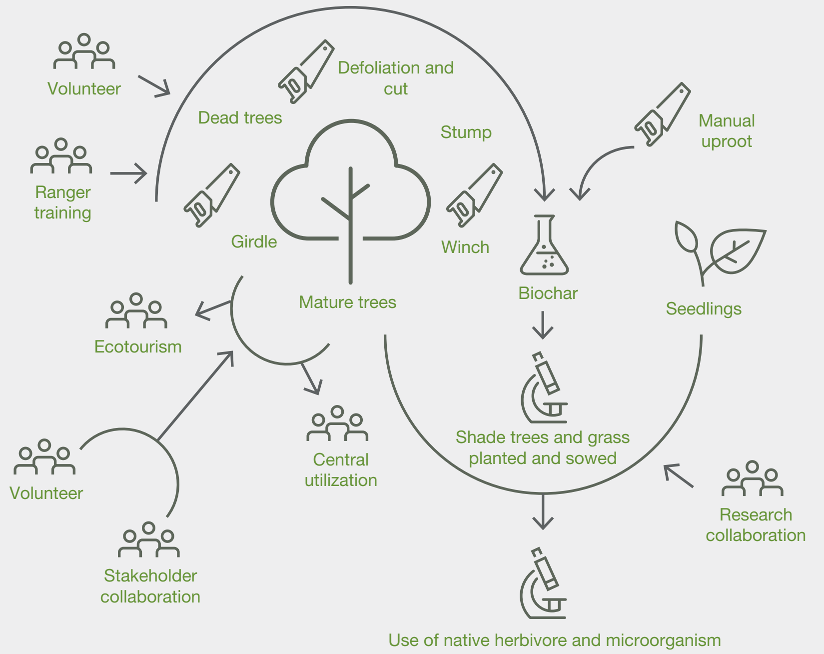


Figure 5 19 **Plan for integration of stakeholder engagement and management strategies for *Vachellia nilotica* (gum arabic tree) management in Baluran National Park, Indonesia.**

This figure shows the suggested optimum method integration of control strategies of *Vachellia nilotica*. Source: Zahra *et al.* (2020), https://doi.org/10.13057/biodiv/d210115, under license CC-BY-NC-SA 4.0.

management includes setting clearly articulated objectives around a future desired state (K. Park, 2004). Adaptive management is learning by doing approach so is contingent on the implementation of a monitoring programme that is able to quantify which action(s) led to changes in the distribution and abundance of invasions, and the ecosystem response and why. The agreed best course of management from a possible suite of actions is then selected and modified in a continuous adaptive cycle of implementing actions, monitoring, learning, and adjustment of new actions to improve the efficiency of management practices (Roux *et al.*, 2011; Zalba & Ziller, 2007; **section 5.3**). Adaptive management can be supported by sequentially considering prioritization based on actual and/or potential impacts, assessing the feasibility and likelihood of success of different control approaches, and clearly defining the goal of the management response (**section 5.2**). This assists in adjusting and selecting the most appropriate management strategy (Lyons *et al.*, 2008). Uncertainties and gaps in information are inherent in the knowledge base upon which adaptive management is applied, and when new information becomes available and scientifically tested, management strategies can be adjusted and improved (K. Park, 2004; **section 5.6.2**). Stakeholders are an integral part of the system, and their full support is a precondition of success. The likelihood of long-term sustainable co-management can be enhanced with a common understanding of the problem, including the responsible management agency, general public and other stakeholders and Indigenous Peoples and local communities (R. T. Shackleton, Adriaens, *et al.*, 2019; R. T. Shackleton, Larson, *et al.*, 2019; **section 5.4.1**).

b) Ecosystem restoration

Invasive alien species not only alter *in situ* ecological community assembly, but also the intended endpoint communities following ecosystem restoration (D'Antonio & Meyerson, 2002; **Chapter 1**, **Box 1.7**; **Chapter 6**, **Table 6.7**, **section 6.7.1**). As such, controlling invasive alien plants has become a significant ecosystem restoration management problem (D'Antonio *et al.*, 2016; Prior *et al.*, 2018; Weidlich *et al.*, 2020). Ecosystem restoration is also an important follow-up to invasive alien species management. Because invasive alien species may hinder the establishment and growth of native species, passive ecosystem restoration (the removal of the invasive alien species) may not be enough, and active ecosystem restoration may be implemented (Brancalion *et al.*, 2019). This may include the use of alternative native species to functionally replace the removed invasive alien plants (Gigon, 2007) or more controversially use of invasive alien species for restoration, when this might be acceptable (e.g., Vimercati *et al.*, 2020). Ecosystem restoration through increased biotic resistance (**Glossary**), can also help prevent colonization of these sites and ecosystems by other invasive alien species that might replace those removed. Legacy effects, where the degradation history of the invaded site or ecosystem including changes in soil nutrients (Nsikani *et al.*, 2018), determines the capacity of the site to self-restore or lead to unexpected consequences following removal of the invasive alien species, need to be understood and managed (Stephens *et al.*, 2009; **Chapter 3**, **section 3.3.5.1**; **Chapter 6**, **section 6.3.3.3**). In some contexts it may be important to ensure restored sites are connected to unrestored sites such as in aquatic restoration situations (Besacier-Monbertrand *et al.*, 2014). The success of ecosystem restoration on sites where invasive alien species are managed also depends on long-term monitoring to understand and manage any further incursions or re-invasions (Trowbridge *et al.*, 2017). A recent global review has shown that non-chemical (mainly mowing and prescribed fire) and chemical (mainly glyphosate) control of invasive alien species was used in 58 per cent and 42 per cent of studies respectively (Weidlich *et al.*, 2020). Decisions on which control method to use are dependent on the growth form of the invasive alien species and resources available for control. The review also found most studies were in temperate deciduous forest and grasslands in developed countries, where chemical control was widely used, whereas in developing countries (low access to technology solutions) where ecosystem restoration has been undertaken used only non-chemical methods. Greater knowledge is needed on how best to manage invasive alien species as part of ecosystem restoration in developing countries (where most high diversity ecosystems occur). A number of guidance documents exist on how to manage the risks of invasive alien species during ecosystem restoration management (UPGE, 2020). As the Indigenous Peoples Local Biodiversity Outlook noted, traditional knowledge can provide contributions to ecosystem restoration in relation to invasive alien species. Incorporating traditional knowledge into ecosystem restoration provides opportunities to strengthen partnerships leading to improved project implementation while increasing ecological viability, social acceptance and economic feasibility (Forest Peoples Programme *et al.*, 2016). See **Supplementary material 5.7** for more details and Indigenous Peoples and local community examples.

5.4.4 Summary tables

Based on the evidence collated in this section we provide three comparative summary tables for these technologies, tools and approaches for a) broad effectiveness of each approach, tool or technology for four different management contexts across the invasion continuum **(Table 5.6)**, b) broad relevance of each technology for application to a given weed, pest or disease type by sector **(Table 5.7)** and c) comparative summary for each technology across management contexts for cost-effectiveness, the time between the application of the technology and some desired outcome/impact and relevance of application at different spatial scales of response or management **(Table 5.8)**. The application of these technologies is limited for marine systems, but where applications have been made these have been discussed.

Table 5 6 **Comparative guide to applicability of decision-support tools and each approach, tool or technology discussed in sections 5.2 and 5.4.**

Assessment categories relate to use contexts discussed in the individual technology specific subsections. The table distinguishes four broad areas of management action associated with the four stages of invasion curve in **Figure 5.1**. The assessment categories are generally relevant (✓), not generally relevant (✗) and some relevance (✗✓), with footnotes providing additional information.

TECHNOLOGY	BROAD AREAS OF MANAGEMENT ACTIONS			
	Surveillance/ Detection	Eradication	Containment	Widespread Control
Decision-support tools				
Qualitative and quantitative decision-support tools	✓	✓	✓	✓
Relevant databases and analytics for management of biological invasions	✓	✓	✓	✓
Surveillance, detection and diagnostics				
Digital data mining – crowdsourcing general surveillance	✓	✓	✓	✓
Sensor-networks and smart traps	✓	✓	✓	✗✓
Screening technologies	✓	✗	✗	✗

Table 5 6

TECHNOLOGY	BROAD AREAS OF MANAGEMENT ACTIONS			
	Surveillance/ Detection	Eradication	Containment	Widespread Control
Surveillance, detection and diagnostics				
Environmental DNA	✓	✓	✓	✗✓
Sentinel surveillance and monitoring	✓	✓	✓	✗✓
Citizen surveillance – data input portals	✓	✓	✓	✓
Earth observation – remote sensing detection	✓	✓	✓[6]	✓[6]
Automated image-based diagnostics and machine learning	✓	✓	✓	✓
Volatile detection technologies	✓	✓	✗✓	✗✓
Pheromone and semiochemical lures[7]	✓	✓	✓	✗✓
Acoustic/ultrasound sensors	✓	✓	✓	✗✓
Point of Care / Lab on a chip, rapid test diagnostics	✓	✓	✓	✓
Track and trace genomics	✓	✓	✓	✗✓
Intervention technologies				
Mechanical & manual approaches	✗	✓	✓	✗✓[8]
Pesticide management of invasive alien animals and plants	✗	✓	✓	✗✓
Robotic technology for targeted management measures	✓	✓	✓	✓
Lethal control of invasive alien vertebrate pests	✗	✓	✓	✓
Fertility control for invasive alien vertebrates	✗	✓	✓	✓
Classical biological control of invasive plants & invertebrates	✗	✗	✓	✓
Sterile insect technique etc.	✗	✓	✓	✗✓
Viral biological control of invasive alien vertebrates	✗	✗	✓	✓
RNA Interference	✗	✓	✓	✗
Genetic-control approaches (including gene-drive)	✗	✓	✓	✓
Adaptive integrated management strategies	✗	✓	✓	✓
Ecosystem restoration	✗	✗	✗✓	✓

6. Remote sensing supporting landscape management and only likely to increase as global broadband internet access become ubiquitous e.g., *via* low orbital satellite constellations
7. Pheromones and semiochemical lures are considered under surveillance, detection and diagnostics but it is reconized that they may be used as an intervention technology (**section 5.5.4**).
8. Generally, these approaches do not provide widespread long-term control except when populations are contained i.e., within an offshore or mainland island context.

Table 5 7 **Comparative guide to the applicability of decision-support tools and technologies discussed in sections 5.2 and 5.4.**

The table distinguishes application of decision-support tools and technologies to invasive alien plants, invertebrates, vertebrates or disease pathogen by sector. Decision-support tools and technologies were assessed with consideration to the contexts in which they are used, as discussed in the individual technology specific subsections. The assessment categories are generally relevant (✓), not generally relevant (✗) and some relevance (✗✓), with footnotes providing additional information. In the context of zoonotic diseases this table refers to diseases transmissible between animals to humans rather than diseases of animal origin largely transmitted between people (e.g., COVID-19).

TECHNOLOGY	Terrestrial invasive alien plants	Aquatic invasive alien plants	Agricultural invertebrate invasive alien species	Environ. invertebrate invasive alien species	Terrestrial vertebrate invasive alien species	Aquatic vertebrate invasive alien species	Plant pathogens	Terrestrial animal pathogens	Aquatic animal pathogens	Zoonotic/ Vector borne pathogens	Marine invasive alien species [9]
Decision-support tools											
Qualitative and quantitative decision-support tools	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Management relevant databases and analytics	✓	✓	✓	✗[10]	✓	✓	✓	✗[9]	✓	✗[9]	✓
Surveillance, detection and diagnostics											
Digital data mining – crowdsourcing general surveillance	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Sensor-networks and smart traps	✗[11]	✗[10]	✓	✓	✓	✓	✗[10]	✓	✗	✓	✗[10]
Screening technologies	✓	✓	✓	✓	✓	✓	✗	✗	✗	✗	✗
Environmental DNA	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Sentinel surveillance & monitoring	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Citizen surveillance – data input portals	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Earth observation – remote sensing detection	✓	✓	✗[12]	✗[11]	✓	✗✓	✓[11]	✗	✗	✗	✗
Automated image-based diagnostics and machine learning	✓	✓	✓	✓[13]	✓	✓	✗	✗	✗	✗	✓
Volatile detection technologies	✓	✗[10]	✓	✓	✓	✗[10]	✓	✓[10]	✗	✓[10]	✗[10]
Pheromone and semiochemical lures	✗	✗	✓	✓	✗	✗	✗	✗	✗	✗	✗
Acoustic/ultrasound sensors	✗	✗	✓[14]	✓[13]	✓	✓	✗	✗	✗	✗	✗✓
Point of Care / Lab on a chip, rapid test diagnostics	✗	✗	✗	✗	✗	✗	✓	✓	✓	✓	✗
Track and trace genomics[15]	✗	✗	✓	✓	✗	✗	✓	✓	✓	✓	✗
Intervention/control technologies											
Mechanical & manual approaches	✓	✓	✗[15]	✗	✗[16]	✗	✗	✗	✗	✗	✗
Pesticide management of invasive alien animals and plants	✓	✓	✓	✓	✓	✓	✗[17]	✗[16]	✗[16]	✗[16]	✗

9. Intervention and control technologies are applied but have so far proved ineffective in marine systems beyond very short-term control.
10. Databases for these sectors do not appear to be well developed.
11. Appear not yet demonstrated as effective for these sectors, but where relevant considered to have potential.
12. Where there is a detectable signal e.g., in the attacked host plant for pathogens and invertebrate herbivores.
13. Only where species are taxonomically defined, which is not always the case.
14. Where noise making.
15. *Via* pan-genomic full genome sequencing which can also track intraspecific genetic variation.
16. Only exceptions are burrowing species like beetle grubs or rabbits.

Table 5 7

TECHNOLOGY	Terrestrial invasive alien plants	Aquatic invasive alien plants	Agricultural invertebrate invasive alien species	Environ. invertebrate invasive alien species	Terrestrial vertebrate invasive alien species	Aquatic vertebrate invasive alien species	Plant pathogens	Terrestrial animal pathogens	Aquatic animal pathogens	Zoonotic/ Vector borne pathogens	Marine invasive alien species [9]
Intervention/control technologies											
Robotic technology for targeted management measures	✓	✓	✓	✓	✓	✓	✓ [10]	✗	✗	✗	✓
Lethal control of invasive alien vertebrate pests	✗	✗	✗	✗	✓	✓	✗	✗	✗	✗	✗ [10]
Fertility control for invasive alien vertebrates	✗	✗	✗	✗	✓	✓	✗	✗	✗	✗	✗
Classical biological control of invasive plants & invertebrates	✓	✓	✓	✓	✗	✗	✗✓ [18]	✗	✗	✗	✗
Sterile insect technique etc.	✗	✗	✓	✓ [10]	✗	✗	✗	✗	✗	✗✓	✗
Viral biological control of invasive alien vertebrates	✗	✗	✗	✗	✓	✓	✗	✗	✗	✗	✗
RNA Interference	✗	✗	✓	✓	✗	✗	✓	✓	✓	✓	✗
Genetic-control approaches (including gene-drive)	✓ [10]	✓ [10]	✓	✓	✓	✓	✗ [19]	✗ [18]	✗ [18]	✗ [18]	✓
Adaptive integrated management strategies	✓	✓	✓	✓	✓	✓	✓	✗ [10]	✗ [10]	✗ [10]	✓
Ecosystem restoration	✓	✓	✓	✓	✓	✓	✗✓	✗	✗	✗	✗

17. Only shows effectiveness for fungal pathogens in agriculture using fungicides no demonstrated effectiveness in native ecosystems invaded by invasive alien pathogens.
18. Rarely effective (Scott, 1995).
19. Genetic-control approaches for disease resistant commercial plants and animals is widely used in agriculture but this is not discussed in **section 5.4.4.2**

Table 5 8 Comparative guide to decision-support tools and technologies discussed in sections 5.2 and 5.4.

This table provides an assessment of decision-support tools and technologies for cost-effectiveness, the time between the application of the technology and some desired outcome/impact and relevance of application at different spatial scales of response or management. Decision-support tools and technologies were assessed with consideration to the contexts in which they are used, as discussed in the individual technology specific subsections. Timeframe of benefit can be: short (quick but effective only in the short-term effective); medium (effective only in the medium term); long (within years of application and providing long-term effectiveness). The assessment categories are generally relevant (✓), not generally relevant (✗) and some relevance (✗✓), with footnotes providing additional information.

TECHNOLOGY	Cost-effectiveness	Timeframe of benefit	Site	Catchment	Region (within country)	Country
Decision-support tools						
Qualitative and quantitative decision-support tools	✓	Short-Long	✓	✓	✓	✓
Management relevant databases and analytics	✓	Medium	✗✓	✗✓	✓	✓
Surveillance, detection and diagnostics						
Digital data mining – crowdsourcing general surveillance	✓	Short	✓	✓	✓	✓

Table 5 8

TECHNOLOGY	Cost-effectiveness	Timeframe of benefit	Site	Catchment	Region (within country)	Country
Surveillance, detection and diagnostics						
Sensor-networks and smart traps	✓	Short-Long	✓	✗	✗	✗
Screening technologies	✓	Short	✓	✗	✗	✗
Environmental DNA	✓	Short	✓	✓	✓	✓
Sentinel surveillance & monitoring	✓	Medium	✓	✓	✓	✓
Citizen surveillance – data input portals	✓	Medium-Long	✓	✓	✓	✓
Earth observation – remote sensing detection	✓	Short-Long	✓	✓	✗✓	✗✓
Automated image-based diagnostics and machine learning	✓	Short	✓	✓	✓	✓
Volatile detection technologies	✓	Short	✓	✗	✗	✗
Pheromone and semiochemical lures[20]	✓	Short	✓	✓	✗	✗
Acoustic/ultrasound sensors	✓	Short	✓	✗	✗	✗
Point of Care / Lab on a chip, rapid test diagnostics	✓	Short	✓	✓	✓	✓
Track and trace genomics	✓	Short	✓	✓	✓	✓
Intervention/control technologies						
Mechanical & manual approaches	✓[19]	Short	✓	✓	✗	✗
Pesticide management of invasive alien animals and plants	✓[19]	Short-medium	✓	✓	✗	✗
Robotic technology for targeted management measures	✓[21]	Short-Long	✓	✓	✓	✗
Lethal control of invasive alien vertebrate pests	✓	Short-Medium	✓	✓	✓	✗
Fertility control for invasive alien vertebrates	✓[22]	Short-Medium	✓	✓	✗	✗
Classical biological control of invasive plants & invertebrates	✓[23]	Medium-Long	✓	✓	✓	✓
Sterile insect technique etc.	✓[24]	Short	✓	✓	✓	✗
Viral biological control of invasive alien vertebrates	✓	Medium-Long	✓	✓	✓	✓
RNA Interference	✓[22]	Short	✓	✓	✗	✗
Genetic-control approaches (including gene-drive)	✓[25]	Long	✓	✓	✓	✓
Adaptive integrated management strategies	✓	Short-Long	✓	✓	✗	✗
Ecosystem restoration	✓	Medium-Long	✓	✓	✗	✗

20. Pheromones and semiochemical lures are considered under surveillance, detection and diagnostics but it is reconized that they may be used as an intervention technology (**section 5.5.4**).
21. Likely to vary on context e.g., land values and/or area of application.
22. Only in contained populations so far without an oral delivery system (not currently available).
23. Where feasibility and success likelihood are high on species by species basis.
24. Where feasibility and success likelihood are high for some invertebrates (sterile insect technique or RNAi) and pathogens (RNAi) only.
25. As not yet field tested so only cost-effective if it works.

5.5 MANAGEMENT STRATEGIES

This section reviews the effectiveness, successes and failures of pathway management (prevention) and species-based (eradication, containment, control) and site- and ecosystem-based (integrated management and restoration) management illustrated with case studies.

5.5.1 Prevention – managing pathways

It is widely accepted that preventing invasive alien species introductions, where possible, is the most cost-effective initial response to managing aquatic and terrestrial biological invasions (Wittenberg & Cock, 2003), but for marine biological invasions it is currently the only efficient option (Hewitt & Campbell, 2007; Galil *et al.*, 2019). The imperatives (**section 5.1.1**), decision-support tools (**section 5.2.2.1**), approaches (**section 5.3.1**) and tools and technologies (**sections 5.4.2**, **5.4.3**) have been addressed in previous sections. This section reviews the effectiveness of implementing prevention and preparedness strategies.

In a terrestrial context, the IPPC and its ISPMs support effective management of most invasive alien species pathways associated with plant trade (**section 5.3.1.1**; Schrader & Unger, 2003; Hedley, 2005). While no formal review has been undertaken on the effectiveness of the IPPC in preventing international movement of invasive alien species, it is widely accepted that these have contributed to significantly reducing unintentional introductions (**section 5.3.1.1**; **Chapter 6**, **Table 6.8**). Nonetheless, many invasive alien species move unaided across contiguous land masses, and are therefore poorly contained by trade controls in this context. The global spread of *Spodoptera frugiperda* (fall armyworm) is a recent example (Tay *et al.*, 2022). Countries within contiguous land masses have been largely ineffective at halting the natural spread of invasive alien species. This is so between jurisdictions in general and political unions (e.g., the United States (Corn & Johnson, 2013) and the European Union (Hulme *et al.*, 2009), trade blocs such as the Association of Southeast Asian Nations (ASEAN; Castriciones & Vijayan, 2020) and Mercado Común del Sur in South America (Southern Common Market, Mercosur; Black & Bartlett, 2020) or international aid and trade initiatives, which may have led to more rapid natural spread of invasive alien species despite any form of curtailment (**Chapter 3**, **sections 3.2.2.3** and **3.2.3**; Liu *et al.*, 2019). Evidence that prevention works for island nations stems largely from reviews of Australian (CSIRO, 2020; Schneider *et al.*, 2020) and New Zealand's (Delane, 2019) national biosecurity systems which also are well established and enacted scrupulously. In any case, a biosecurity system is worth the investment especially for all island nations. The most obvious metric of the effectiveness of these systems is that establishment rates of new invasive alien species are near zero for invasive alien vertebrates and animal diseases and largely constant in invasive alien plants, invertebrates and plant pathogens (Bailey *et al.*, 2020; CSIRO, 2020; Hulme, 2020b; A. W. Sheppard & Glanzing, 2021) and predicted to remain so for some groups rather than continuing to grow at fast rates in most other countries (Seebens *et al.*, 2017). One clear example is the processes put in place across Australasia (Australian Government, 2021b) and New Zealand (Ministry for Primary Industries, New Zealand, 2021) for the pathway management of *Halyomorpha halys* (brown marmorated stink bug) as a very high priority threat to the agricultural sectors of both countries intercepted in significant numbers every year. Both countries have put in place a systems-based pathway management approach that is causing a decline in the numbers of interceptions (Australian Department of Agriculture, 2019). This suggests that systems-based approaches (a series of risk mitigation interventions along the supply chains) are an effective way of managing pathways for terrestrial invasive alien species (**section 5.4.3.1**; van Klinken *et al.*, 2020). Effective prevention is also supported by effective intelligence gathering on changing and future trade and pathway risks and effective national preparedness (**sections 5.2.2.4** and **5.3.1.1**). Compared to this, most developing countries are not so successful in implementing effective pathway management because of outdated biosecurity systems or a lack of diligence and capacity in enacting the regulations contained therein (Gupta & Sankaran, 2021).

WOAH Standards aim to prevent movement of animal diseases (**Chapter 6**, **Table 6.8**), however, many of these have been poorly implemented internationally as seen by the current pandemic spread of African Swine Fever particularly through Asia (Dixon *et al.*, 2020) and now into the Americas. Pandemic spread of African swine fever is largely due to it being a highly contagious haemorrhagic viral disease spread through wild, domesticated, dead and live pigs, contaminated feed and fomites and trough pork products (Ward *et al.*, 2021).

Pathway management is currently the only effective option for preventing marine biological invasions (Hewitt & Campbell, 2007; Galil, McKenzie, *et al.*, 2019; **section 5.5.3** and **Figure 5.1B**) as recorded number and spread of marine invasive alien species are increasing with few exceptions (Bailey *et al.*, 2020). Ballast water has been an important dispersal pathway of invasive alien species since the late 1800´s. The first major comprehensive review of the biology of ballast water was in 1985 (Carlton, 1996), based on quantitative data only available from mid 1980s (**Chapter 3**, **section 3.2.3.1**). Since then, ballast water research grew exponentially with 400 publications from

1955 to 2013 (Bailey, 2015). Several countries started to regulate ballast water management from 1989 at regional or national levels (Bailey, 2015; Hewitt *et al.*, 2004). Today, the BWM Convention has been adopted by the International Maritime Organization (IMO) and entered into force in 2017 to help prevent the spread of potentially harmful aquatic organisms and pathogens in ships' ballast water. Adoption the BWM Convention requires ships to manage their ballast water so that aquatic organisms and pathogens are removed or rendered harmless before the ballast water is released into a new location (IMO, 2004). The efficacy of the ballast water management (for details see **section 5.4.3.1**) has only been tested in a few countries, and more long-term studies are needed to understand its efficacy on preventing new species introductions. Bailey *et al.* (2011) evaluated the efficacy of ballast water tank flushing to reduce the introduction of invasive alien species and found a declining rate of detections of them in the Great Lakes. It is estimated that nearly 70 per cent of the marine invasive alien species established worldwide were introduced *via* biofouling (Hewitt & Campbell, 2010). In countries and regions where biofouling regulations exist, the efficacy of these in managing invasions is not well understood given that regulations are recent and were implemented only in a few countries and regions (K. R. Hayes *et al.*, 2019).

Regulations covering several biofouling management strategies were reviewed for New Zealand by Morrisey and Wood (Morrisey & Woods, 2015) and include:

- In-water cleaning and capture systems, which, according to a recent study, can reduce biofouling by 82-94 per cent (Tamburri *et al.*, 2020). However, evaluation of different factors affecting system performance (vessel parameters – type, design, coating; environmental parameters – water visibility, currents, winds, water quality; and in-water cleaning system design and operations – operator experience, debris capture, frequency and rate of operation, etc.) is needed (Tamburri *et al.*, 2020).

- Manual cleaning after beaching the vessel, which can be effective in certain conditions and when vessels are small or stable (e.g., recreational vessels; Castro *et al.*, 2020; and also Government of Canada, 2021).

- Encapsulation treatment (with seawater alone or added with acetic acid or chlorine in small recreational vessels). Tests showed that treatment with seawater for 5 days was enough for eliminating 100 per cent of the organisms (Keanly & Robinson, 2020), but the additives highly reduced kill time (Atalah *et al.*, 2016; Forrest *et al.*, 2007; Morrisey *et al.*, 2016; Roche *et al.*, 2015).

An adaptive or systems-based management approach is needed when applying these different management methods as efficacy is subject to local regulatory, logistical and environmental conditions which differ from one region to another, even within the same country.

Some Indigenous Peoples and local communities apply local quarantine measures prohibiting the transport of certain species that are not used in their cultural practices and customs. Elders provide awareness raising, education and capacity-building passing on oral knowledge from one generation to the next. Teso and Bukusu/Bagisu Kenya-Uganda transboundary customs and cultural practices transfer knowledge during festivals and ceremonies (Angujo, 2015; Barasa, 2012).

5.5.2 Surveillance, detection and monitoring

The main purpose of surveillance is to detect or ensure the absence of new invasive alien species or disease incursions at the border and onshore on time to attempt eradication (**section 5.3.1.2**). Failure to detect incursions rapidly is the major factor limiting the effectiveness of eradication programmes **(Figure 5.1)**. **Box 5.14** shows how surveillance can play an important role in global biosecurity systems, by helping early detection of invasive alien species which have led to successes in eradication of newly established populations (Gerda, 2021). Monitoring of established populations and risk analysis are also important to understand invasiveness to support management actions (Jarrad *et al.*, 2015). Surveillance systems are rarely perfect and this is one of the reasons why eradications can be hard (Rout *et al.*, 2009). In South Africa, surveillance was used to detect and manage populations of *Asphodelus fistulosus* (onionweed) as part of an eradication programme (Jubase *et al.*, 2019). Active specific surveillance in conjunction with general public surveillance (using awareness raising and solicited reporting) were undertaken. Detected populations were treated and monitored over a four-year period to assess the feasibility of eradication leading to effective management. A number of studies also demonstrate that effective surveillance designed for detection at low prevalence or incidence maximizes the effectiveness and lowers the costs of eradication programmes (Kalwij *et al.*, 2014; Pluess, Jarošík, *et al.*, 2012; Reaser, Burgiel, *et al.*, 2020; Simberloff, 2003). Some surveillance programmes seek to optimize post eradication detection to ensure management success is maintained (Epanchin-Niell *et al.*, 2014). See **Supplementary material 5.9** for further details. A case study which also demonstrates the effectiveness of structured surveillance is the Australian red imported fire ant eradication programme **(Box 5.14)**.

A risk-based approach to surveillance can identify priority invasive alien animal and plant and diseases and provide a basis for resource allocation (A. R. Cameron, 2012;

Box 5 14 **The New Zealand National Invasive Ant Surveillance Programme as an example of early detection for successful eradication of invasive ants.**

Established in 2003, following a successful eradication of *Solenopsis invicta* (red imported fire ant) incursion at Auckland International airport in 2001, New Zealand's National Invasive Ant Surveillance programme ensured surveillance of shipping ports, airports and international cargo facilities. Since 2002, approximately 418 baited traps with food attractant deployed over 18 sampling seasons in the programme have recorded invasive ant species. Most of these detections were of ants from newly established nests (Peacock *et al.*, 2019), eradicated under "urgent measures" soon after detection. In 2019, there were 19 ant detections, of which 11 were associated with established ant nests (Peacock *et al.*, 2019) and were eradicated for 29,000 New Zealand dollars (NZ$). The ant surveillance programme costs approximately NZ$ 500,000 per annum. The cost-benefit ratio of continued surveillance is high compared to NZ$ 8.6 million spent over 3 years to eradicate the red imported fire ant in Whirinaki, Napier, New Zealand from 2006 to 2009 (Gerda, 2021, 2021). The cost of living with red imported fire ant in New Zealand without the programme has been estimated at NZ$ 318 million per annum (Anon, 2001; **Figure 5.20**).

Figure 5 20 **Invasive alien ant trap used in New Zealand as part of a surveillance programme.**

Photo credit: Dr Paul Craddock – under license CC BY 4.0.

Hoinville *et al.*, 2013; Oidtmann *et al.*, 2013). For example, Grace *et al.* (2020) demonstrated an effective risk-based surveillance system for bluetongue virus built on multiple components to know where and when to target surveillance to ensure disease freedom. The risk-based surveillance components consisted of international disease monitoring and post import testing of livestock from high risk areas, and arrival and establishment of the vector, *Culicoides* spp. (biting midges; Grace *et al.*, 2020). Syndromic surveillance of disease status based on clinical signs or other data has been effective for picking up changes in the incidence of disease (Hoinville *et al.*, 2013). This was useful for detecting the first sign of Bluetongue disease serotype 8 in North Western Europe in 2006 (Elbers *et al.*, 2008). Passive surveillance in animal health is when farmers report potential diseases to their veterinarians and the information is collated and reported (del Rocio Amezcua *et al.*, 2010). In Tanzania's animal health system, disease reporting is mostly passive. Clinical observation data from 13 primary sources (mainly livestock farmers, abattoirs, livestock markets, etc.) provide an overall picture of animal health (George *et al.*, 2021).

The international plant sentinel network (**Supplementary material 5.3**) is an effective early warning system for new and emerging pest and pathogen risks through a global network of National Plant Protection Organizations, scientists, botanic gardens and arboreta around the world (Barham *et al.*, 2016). This network aims to report plant health issues safeguarding susceptible plant species worldwide. CABI's PlantwisePlus programmes[26] is another effective plant health support system for smallholder farmers in developing countries across Africa, Asia and Latin America (Cameron *et al.*, 2016). Farmers bring pests or damaged crops for identification and receive pest and disease management advice, contributing to the early detection of new plant pests (Migiro & Otieno, 2020).

26. https://www.plantwise.org

ProMED-mail, the programme for monitoring emerging diseases, also reports on outbreaks of human infectious diseases and monitors diseases of agricultural importance in plants and animals using the internet to mine information sending online reports to subscribers (Yu & Madoff, 2004). Reports are validated by expert moderators. Other effective early warning systems include PestLens (US Department of Agriculture), European and Mediterranean Plant Protection Organization (EPPO) alert list and reporting service for member countries. The NAPPO Phytosanitary Alert System provides a similar service for Canada, United States and Mexico. The IPPC provides a similar service for all national plant protection organizations (Noar *et al.*, 2021).

Most successful examples of priority quarantine pest and invasive alien species surveillance are from developed countries (Mphande, 2016). Elsewhere, such surveillance is under-reported or not practiced on a regular basis. Most one-off surveillance and monitoring surveys of invasive alien species in the Pacific region are not formally published. In contrast, the Pacific Invasive Ant Toolkit provides advice on biosecurity and surveillance for invasive ants (Gruber *et al.*, 2016) rapidly spreading across the Pacific (McGlynn, 1999). General surveillance requires diagnostic and investigative support services (Froud & Bullians, 2010). New Zealand's general surveillance system covers animal, plant and environment health (Bleach, 2019; Tana, 2014) with a National Call Centre emergency phone service supported by experts, laboratory diagnostics and investigators. The results are published online quarterly (Ministry for Primary Industries, New Zealand, 2020).

A mobile phone-based identification tool designed jointly by the New Zealand Government and the Māori community called Find-A-Pest has improved public passive surveillance reporting levels (Pawson *et al.*, 2020) such that 95.5 per cent public identifications were correct with a 56.1 per cent successful hit record for high priority species profiled on the factsheets embedded in the Find-A-Pest application. General surveillance has also been successful in New Zealand for a range of marine species by different communities: the ascidian *Eudistoma elongatum* reported by marine aquaculture (Smith *et al.*, 2007), the *Charybdis japonica* (lady crab) by commercial fishers (Smith *et al.*, 2003) and Ostreid herpesvirus Type 1 (OsHV-1) from noticed mortalities in juvenile oysters by the industry (Bingham *et al.*, 2013). Other examples include a surveillance programme developed to detect invasive alien mosquitoes (Mosquito Alert, 2021) and FAMEWS, a mobile app used for monitoring and early detection of fall armyworm (FAO, 2021)

Environmental DNA metabarcoding (**section 5.4.2.1d**) is being used in marine systems in some countries but it is an expensive tool, and sequence databases/libraries are being developed for many species at a slow rate. Zaiko et al (2015) found it was five times more effective than classical morphological analyses in detecting invasive alien species in plankton samples in the Baltic Sea, although accuracy can be a concern (Ricciardi *et al.*, 2021; **Chapter 6**, **Box 6.19**).

Activities and knowledge systems of some Indigenous Peoples and local communities' effectively support surveillance (Ingold, 2000). The Mayan lobster diver-fishers were the first to detect *Pterois* spp. (lionfishes) in the Parque Nacional Arrecife Alacranes (southern Gulf of Mexico; López-Gómez *et al.*, 2014). Some Indigenous Peoples and local communities have robust invasive alien species detection systems **(Box 5.15)** which have many similarities with internationally recognized systems (ICIPE, 2018; Shine, 2005). In some communities, the council of elders for a given region monitors and evaluates the entire ecosystem situation and gives reports during the meeting of Indigenous Peoples and local communities. The council of elders works in harmony with the People's culture and customs (Aiken *et al.*, 2015).

Effective surveillance by Indigenous Peoples and local communities of native ecosystems in Kimberley, Northwest Australia is part of hunting, fishing and gathering, and the

Box 5 15 **Surveillance and management of invasive alien species by Indigenous Peoples and local communities – A case study of The Bukusu community in Kenya.**

The Bukusu community notifies an elder when a new plant species is first found in their environment. A council of elders confirms the detection and quarantine is imposed. A date is then set for a ritual ceremony to determine whether management of the plant should proceed. At the ceremony, a sheep is slaughtered at the detection site and its stomach contents together with samples of plant shoot (called *Lufufu*) are mixed in water which the elder places on and around the plant while some ceremonial statements are made. On the 3rd day the *Lufufu* leaves are checked to see if they are dry, following which the plant is uprooted and burnt. If the leaves are still healthy the plant is considered good for the native ecosystem, given a local name and its uses and applications are defined based on similar local plant species. If a new animal species is detected (whether *Esang'i*- the eaten animal species or *Esolo*- a non- eaten animal species) the council of elders identify its foot prints and a child is given a mixture of *Kulandula* plant to put in the foot prints as the elders curse the animal never to return since its effects to the native ecosystem, economy and livelihoods are not known (Wanzala *et al.*, 2012).

reporting of new species is encouraged. Barter trade is strictly conducted only with known fauna, flora and/or minerals. This knowledge of fauna, flora and/or minerals is held by elders by memory and trust and is passed on from generation to generation by word of mouth (Wanzala *et al.*, 2012; Weir & Duff, 2015). Combining both Indigenous and scientific knowledge has improved the understanding on the spread of invasive alien species among local communities. The observations of forest-dwelling Soliga community of South India on *Lantana camara* (lantana) invasion have helped to better understand the process of invasion and plan future management of the species (Sundaram *et al.*, 2012). The Māori Tuawhenua community of Ruatahuna in New Zealand has developed extensive knowledge systems around endemic biodiversity and forest health perceiving changes in the forest and introduced invasive alien species over 65 years (Lyver *et al.*, 2016). See **Supplementary material 5.9** for more examples of effective surveillance. Although surveillance for invasive alien species is a regular process in the developed countries, it is seldom conducted in some of the developing countries for want of updated technical know-how and resources (Gupta & Sankaran, 2021).

5.5.3 Eradication

Successful eradication of an invasive alien species is underpinned by effective surveillance, detection and extirpation of all individuals of the species, which is supported by efficient methods to remove all pre-reproductive individuals (**section 5.4**), good decision-support systems (**section 5.2**) and sustained public and financial support. Sustained monitoring can ensure that there are no new recruitments (Genovesi, 2001; Rejmanek& Pitcairn, 2002; Lehtiniemi *et al.*, 2015; Simberloff, 2020), and the success of any eradication programmes depends on adequate resourcing until all the individuals are removed (Simberloff, 2009). In general, successful eradication programmes that interacted with human activities were achieved with strong stakeholder support through effective engagement, education and communication (Myers *et al.*, 1998; Simberloff, 2003). It is also important to evaluate in advance the conditions which may thwart an eradication programme – for example, an eradication attempt of *Prosopis juliflora* (mesquite) in Ethiopia failed due to lack of resources (Rettberg, 2010; **section 5.6**).

A review of eradication programmes of invasive alien plants conducted by Rejmanek & Pitcairn (2002) concluded that management of populations spread across habitats greater than 1000 hectares is very unlikely to be successful, especially if the target has high spread rates (e.g., *Lantana camara* (lantana); Ranjan, 2019) or seedbanks are hard to detect. It is difficult to eradicate invasive alien plants compared to invasive alien vertebrates (Robertson *et al.*, 2019). Moreover, successful eradications of invasive alien plants were of those which infested smaller areas than those of invasive alien vertebrates (Rejmanek & Pitcairn, 2002; Robertson *et al.*, 2019). At a global scale, several programmes have been implemented since the 1970s to eradicate invasve alien forest insects, with most documented examples proving successful (Brockerhoff *et al.*, 2010; Liebhold *et al.*, 2016; Liebhold & Kean, 2019; Tobin *et al.*, 2014). The cost of forest pest eradication programmes increases with the size of the area affected (Brockerhoff *et al.*, 2010; **Box 5.16**; **Supplementary material 5.10** for more examples).

An eradication programme of *Oryctolagus cuniculus* (rabbits) in Tierra del Fuego (Argentina), which disturbed soil and threatened native species, was legally challenged by animal rights supporters (CADIC-CONICET, 2020; **section 5.6.2**). A mosquito surveillance programme was set up in New Zealand in 1998 in response to the infestation of *Aedes camptorhynchus* (southern saltmarsh mosquito), and its eradication programme which lasted over 10 years costed NZ$ 70 million (Kay & Russell, 2013; **Supplementary material 5.10**). In 2018, the programme detected a few mosquito *(Culex sitiens)* larvae in marshland (McGinn & Disbury, 2019), and a bacterium (*Bacillus thuringiensis israelensis* (Bti)) that kills the mosquito larvae was used as treatment. Three rounds of aerial spraying of Bti were carried out across 5 km from the initial detection sites to eradicate the mosquito. Subsequent surveillance revealed no further infestation of the mosquito.

Reinvasion risk also needs to be addressed in eradication programmes (Pyšek *et al.*, 2020; Spatz *et al.*, 2022) through both natural (e.g., long-distance flights) and anthropogenic (i.e., human-assisted) pathways (Harris *et al.*, 2012). Eradication of *Didemnum vexillum* (carpet sea squirt), a widespread colonial coastal species in western Europe, North America and New Zealand affecting shellfish farms and submerged structures (McKenzie *et al.*, 2017), was attempted in Shakespeare Bay (about 1 km^2), New Zealand (costing NZ$ 650,000) and Holyhead Harbour, Wales, United Kingdom (costing GBP 350,000) (Galil *et al.*, 2019). Approaches included exposing the colonies to desiccation, chemicals, freshwater and physical removal (Rolheiser *et al.*, 2012), but recolonization occurred soon after the eradication efforts stopped. This is a common feature of eradication attempts targeting marine invasive alien species (Galil *et al.*, 2019; McKenzie *et al.*, 2017). Long-term monitoring of all small infestations after eradication is critical in marine systems (Pluess, Cannon, *et al.*, 2012). The eradication of *Carcinus maenas* (European shore crab) was attempted in South Africa using different management techniques including traps, crab condos, diver collections and sediment dredging. However, after one year, crabs were still present and numbers increased as soon as the eradication efforts ceased (Mabin *et al.*, 2020).

Box 5 16 Case study: Successful eradiation of an invasive scale insect in Kerala, India.

Figure 5 21 ***Ceroplastes cirripediformis* (barnacle scale) on a host plant.**

Photo credit: Dr. T. V. Sajeev – under license CC BY 4.0.

Ceroplastes cirripediformis (barnacle scale; **Figure 5.21**) is a highly polyphagous scale insect that causes negative impacts to host plants belonging to 119 genera in 63 families in over 32 countries (García Morales *et al.*, 2016). It sucks the sap of host plants and excretes honey dew resulting in the formation of coal smudge on the affected plant parts. The barnacle scale was identified as an invasive alien species by the Centre for Agriculture and Biosciences International (CABI) in 2017 since it causes significant damage to host plants in its invaded range. It was first recorded in India in 2021 (Joshi *et al.*, 2021). The Nodal Centre for Biological Invasions, Kerala Forest Research Institute, India issued public notices to detect its distribution in Kerala state and it was spotted at one site each in Dhoni and Parali villages in Palakkad District. The host of the insect was a *Passiflora* sp. (passionflower). The identity of the insect was confirmed using molecular methods. Since its spread was very isolated, rapid control was attempted by removing and burning the infested stems and killing the insects at the spot. No new outbreaks were recorded during a 8-month post-eradication period (Swathy, 2021). Surveillance for the insect is being continued.

A global analysis of 173 eradication campaigns in anthropogenic habitats involving 94 species of invertebrates, plants and plant pathogens showed that only 50.9 per cent of the programmes were successful (**section 5.6.1.1**). Both location- and context-specific factors were important for success of eradication, while species-specific characteristics were of minor importance. Invaded areas smaller than 5000 ha had more than 80 per cent of successful eradication probability in man-made habitats (Pluess *et al.*, 2012). It is important to prioritize sites (such as protected areas) for targeting eradication (**section 5.3.2**). Lower success rates were recorded from natural or semi-natural habitats than man-made habitats where success was comparatively more likely due to high economic impacts (**Chapter 4**, **Box 4.13**) and the resultant greater commitments. Eradication success can be ensured with cross-border collaboration and greater cooperation amongst nations (Pluess *et al.*, 2012).

Vertebrate eradication programmes on islands have been particularly successful, especially with rodents (Howald *et al.*, 2007; Spatz *et al.*, 2022), with success numbers increasing exponentially since 1980 (Towns *et al.*, 2019; Carrion *et al.*, 2011; Robertson *et al.*, 2019). Details on successful eradication of invasive alien species on islands can be found in the DIISE **(Table 5.4)**. Recent data show that the success rate was 88 per cent from 1,550 attempts on 998 islands during the last 100 years (Spatz *et al.*, 2022). These successes have been attributed to isolation and small surface area of the islands (Simberloff, 2001). With improved baiting technology (**section 5.4**), eradications were also possible on larger islands which was considered impossible a decade ago (Veitch *et al.*, 2011). This led to targeting human-inhabited islands and continental settings (Malmierca *et al.*, 2011; Zabala *et al.*, 2010; Glen *et al.*, 2013). Roberston *et al.* (2017) found that twelve of fifteen (80%) large-scale mammal removals from Northern Europe since 1900 were successful within defined management boundaries (mean area 2,627 km^2). As such, most programmes were mostly not aimed at eradication from large land masses.

Detailed information on successful eradication programmes (i.e., rate of removal of individuals and techniques applied to achieve results) is generally limited and even less information is available on unsuccessful attempts (Roy *et al.*, 2009; Simberloff, 2020; **section 5.6.2**). In this situation, adaptive management is the most effective approach to eradication, especially when there are data gaps and uncertainty on how best to continue the programme based on early results. A successful example is the removal of *Capra hircus* (goats) from Santiago Island **(Box 5.17)**.

Information on the costs of eradication programmes are necessary to evaluate the economically optimal strategies, however, cost-benefit analyses usually used to evaluate the feasibly of management plans are not frequently published (Pluess, Cannon, *et al.*, 2012). While eradication programmes can only be achieved with access to high immediate costs, they are generally cheaper than long term and permanent control costs and impacts (Bomford & O'Brien, 1995). The eradication of well-established population of *Myocastor coypus* (coypu) through trapping in Great Britain is another success story of eradication. An 11-year campaign (1981-1992) at a total cost of EUR 5 million included dynamic estimate of remaining populations which helped to understand trapping effects on coypu and the trap numbers (Gosling & Baker, 1989; Panzacchi *et al.*, 2007). This programme is comparable to the long-term coypu control programme in Italy which costed EUR 14 million over six years (Panzacchi *et al.*, 2007). In most eradication programmes, the costs of removing individuals escalates greatly based on the fact that the fewer the individuals that are left, the final (remaining) individuals are harder to find. In the eradication of *Cyprinus carpio* (common carp) from Tasmania, Australia, it took just a few years to reduce carp numbers down to a few breeding females, but it took another ten years to track and remove the final individuals, which has only been successful in one of two lakes 25 years after the decision to eradicate was made. Complete success, therefore, has not yet been achieved (Yick *et al.*, 2021).

The costs of eradication can be very high when eradication is only considered as an option at the point when the negative impacts due to the species become visible (Genovesi, 2001). Several invasive alien species projects funded by the Global Environment Facility have focussed on eradication efforts. But, in many cases these were not cost-effective (GEF, 2007; **section 5.3.2**). The costs of multi-species eradication programmes can be lower than eradicating individual species if eradication approaches can simultaneously remove multiple species or the removal of some species facilitates removal of others. Such projects targeting eradication of

Box 5 17 Eradication of goats on Santiago Island, Ecuador.

The large-scale eradication of *Capra hircus* (goats) from Santiago Island, Galápagos Islands, Ecuador **(Figure 5.22)** is an excellent example of successful island eradication. Over 79,000 individual goats were removed from over 58,000 ha in 4.5 years (2001-2005) at a cost of United States Dollar (US$) 6.1 million. This adaptive management programme included ground hunting using specialized techniques, aerial hunting by helicopter, and the use of sterilized Judas (tagged goats used to find other goats through social behaviour; **section 5.4.3.2**) and Mata Hari (females with hormone implants) goats to find and remove the remaining individuals. Methods were constantly revised and adjusted. Different hunting methods were integrated, and hunting efficiencies and escape rates constantly evaluated, contributing to the success of the programme at reduced costs. Removal of the last goats costed $2 million, while the monitoring costs to confirm eradication was $467,064 (Cruz *et al.*, 2009).

Figure 5 22 ***Capra hircus* (goats; cabras in Spanish) invading Santiago Island, Ecuador.**

Photo credit: Heidi Snell/CDF – under license CC BY 4.0.

multiples species in multi-sites have been proven to be cost-effective. For example, in the Archipelago of French Polynesia, the project cost for eradication of various mammal species from six islands was only EUR 1.4 million while the total cost of actions on each island separately was estimated to be EUR 4.6 million. Savings were made on fixed costs such as the costs of helicopters, transport and staff travel (Griffiths *et al.*, 2019; **Box 5.8**).

A clear idea on the size of the area invaded is a pre-requisite to ensure the success of all eradication plans. This is exemplified by the success of species eradication from islands since the area is often smaller compared to large land masses. On large land masses, defining the extent of an invasive alien population may be compounded by the presence of multiple populations, especially when the populations are inter-connected. It is, therefore, essential to understand the meta-population context of a species targeted for eradication, which will help planning the programme and ensuring its efficacy (Robertson *et al.*, 2019). A lack of this understanding given limited resources, is why most culling programmes are ineffective and unsuccessful in the long-term (**section 5.4.3.2**). The ongoing eradication programme of *Oxyura jamaicensis* (ruddy duck) from Europe covers 1,535,509 km^2 requiring participation and investment from several countries (Robertson *et al.*, 2015). The cost of eradication decreases slightly in proportion to the area targeted but there is an island size limit above which eradication may not be successful (Brockerhoff *et al.*, 2010; Robertson *et al.*, 2017, 2019; **Figure 5.23**). In general, the eradication cost per area seems to be similar for both islands and mainland programmes. This example indicates that large scale eradications can be successful.

Aquatic eradication programmes are more frequent in freshwater than marine ecosystems (Simberloff, 2021). In freshwater systems, such programmes are generally restricted to small rivers and lakes and within enclosed bays. Examples include eradication of freshwater bass (*Micropterus* spp.) in a small river in South Africa (O. L. F. Weyl *et al.*, 2013) and programmes targeting species of invasive alien aquatic plants that applied different strategies (Simberloff, 2021). Eradication programmes with invertebrates and small taxa in aquatic ecosystems are less known and have poor success rates given the complex nature of these environments for implementing management procedures and the lack of visibility.

Evidence suggests that there have been no fully successful eradication programmes for well-established invasive alien species in marine ecosystems (Galil *et al.*, 2019). Where

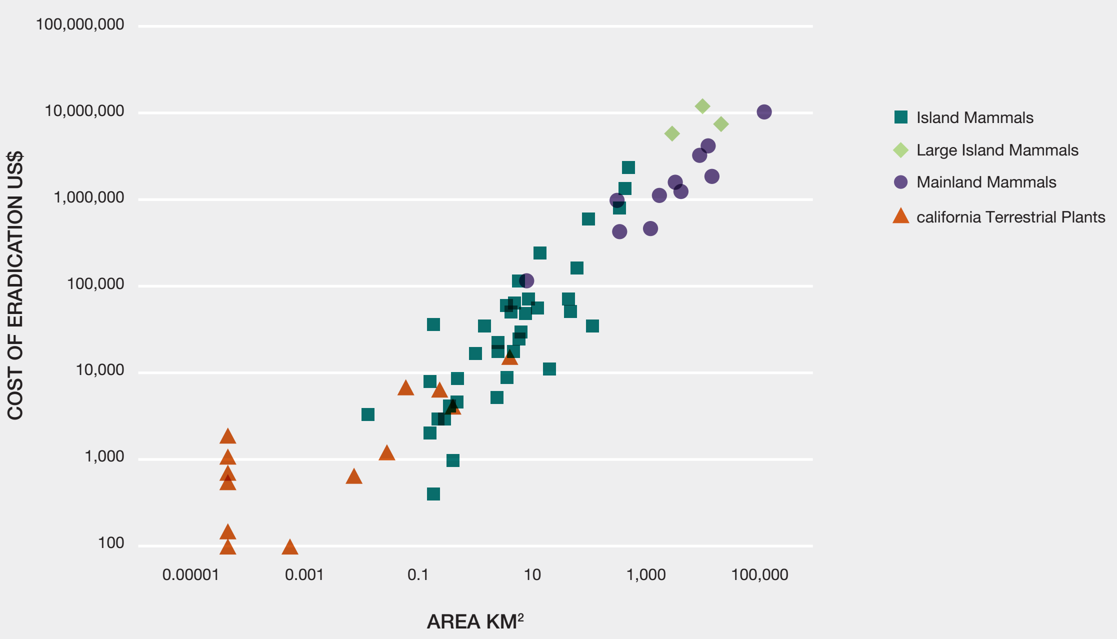


Figure 5 23 **Successful eradication programmes; the relationship between area (km^2, x axis) of successful eradication campaigns and eradication programme cost (y axis).**

Square symbols are island mammal eradications (Martins *et al.*, 2006); circles are mammal eradications from larger land masses in Northern Europe (Robertson *et al.*, 2017); diamond symbols are examples/predictions of large-scale mammal eradications from islands (Cruz *et al.*, 2009; Parkes *et al.*, 2014; Piertney *et al.*, 2016) triangles are plant eradications from California (Rejmanek & Pitcairn, 2002). Source: Robertson *et al.* (2019), https://doi.org/10.2305/IUCN.CH.2019.SSC-OP.62.en, under license CC BY-NC 4.0.

they have been attempted, targets have been restricted to very small initial populations, to sessile biota and small areas. An eradication programme against *Mytilopsis sallei* (Caribbean false mussel), which was discovered in 1999 by local divers, occurring in large densities in three marinas in the port of Darwin in Australia used liquid chlorine and copper sulphate. This killed the mussels and other marine life (Willan *et al.*, 2000; F. E. Wells, 2019). In 2000, *Caulerpa taxifolia* (killer algae) was discovered in a lagoon and then in a harbour in California, United States. A rapid response was activated since this species was included in the Federal Noxious Weed List in 1999. Algal beds were treated with liquid chlorine and a monitoring programme continued and by 2005 the species was considered fully eradicated (Anderson, 2005). In both cases, eradication was possible because the managed area was small, and eradication was carried out soon after locating the species. It is critical to improve eradication programmes in marine ecosystems during the early detection stage, as most attempts at eradicating or containing invasive alien species have been ineffective so far (Galil *et al.*, 2019). Thorough knowledge of each system is needed to avoid failures and non-target impacts which will potentially degrade the ecosystems further and the high costs involved for eradication (Grosholz *et al.*, 2021).

Social aspects of eradication

Successful eradication can lead to ecological and social benefits. In the Seychelles islands, where natural resources have supported the tourism industry (see **section 5.3.2** on management in protected areas), eradication of invasive alien plants and vertebrates, and subsequent reintroduction of native species, has improved tourism, benefiting local people (Samways *et al.*, 2010). In North America, clearance of the invasive shrub *Lonicera maackii* (Amur honeysuckle) altered the behaviour of *Odocoileus virginianus* (white-tailed deer) and its disease vector parasite *Amblyomma americanum* (lone star tick), reducing the risk of vector-borne diseases in humans (Allan *et al.*, 2010). In the arid and semiarid climates of western United States, eradication of the invasive alien plant *Tamarix* sp. (tamarisk) from riparian areas, where it depleted water availability and increased river sedimentation, led to large social and economic benefits to municipalities, farmers, the hydropower industry and fishermen and reduced flood damage in invaded areas (Zavaleta, 2000). On a local scale, therefore, eradication success can lead to increased good quality of life when targeting invasive alien species causing significant negative impacts on economic wellbeing, human health and access to natural resources. Where invasive alien species have commercial, cultural or spiritual value, however, eradication is unlikely to be acceptable (Kelsch *et al.*, 2020; Oppel *et al.*, 2011; **Box 5.18**). The eventual abandonment of *Bubalus bubalis* (Asian water buffalo) eradication in Northern Australia, where the animals were valued by the Indigenous Peoples and local communities, is an example (Ridpath & Waithman, 1988). Where Indigenous Peoples and local communities use invasive alien species for practical, cultural and spiritual purposes eradication could lead to negative consequences on these communities (Atyosi *et al.*, 2019; Haregeweyn *et al.*, 2013; Maldonado Andrade, 2019; **section 5.3.1.3**; **Chapter 1**, **section 1.6.7.1**; **Chapter 4**, **section 4.6.4**).

5.5.4 Containment

Containment is a strategic option to prevent establishment, multiplication and spread of an invasive alien species outside a specific area, often when attempts at eradication become unsuccessful or abandoned (Grice *et al.*, 2012, 2020). Containment aims at delimiting the spread of a species through various management measures though, at times, certain environmental factors may also restrict its spread. This method is often used to manage the spread of invasive alien plants. However, "slowing the spread" is also an option for managing invasive alien pests (Sharov *et al.*, 2002; Sharov & Liebhold, 1998). When opting for containment, resources may be allocated to reduce propagule pressure in

Box 5 18 **Local eradication of cacti *Opuntia* sp. (pricklypear) improves good quality of life in Madagascar.**

Opuntia sp. cacti from South America was first introduced into Madagascar as a defence barrier in the 1700 (Binggeli, 2003). Some species in this highly invasive alien genus are also beneficial providing fodder and some have medicinal properties (Shackleton *et al.*, 2017). Those species with fodder value quickly became a crucial resource for local pastoralists in Madagascar (Kaufmann, 2004), allowing them to have larger herds than the "natural" environmental capacity would allow (Middleton, 1999). The cacti also provided food and water for local communities during dry season. However, range expansion of dense thickets of the cacti reduced land available for crops and native bushy plants (Binggeli, 2003). When *Opuntia* was successfully controlled in southern Madagascar through biological control using *Dactylopius* spp. (cochineal insects), positive outcomes were also achieved that benefitted people in the central highlands. However, loss of *Opuntia* severely affected livelihoods of the pastoralists, who depended on it for food and fodder during droughts which led to migration from the area (Binggeli, 2003; Shackleton *et al.*, 2011).

the zone dominated by the species and in the buffer zone to delimit long-distance dispersal (Grice *et al.*, 2013).

Grice *et al.* (2013) suggested that, for invasive alien plants, containment can be considered as a choice in two main contexts: 1) where an invasive alien species is also a commercially valuable species which can be exploited for that purpose and managed and 2) for a species with no commercial value especially when it has not fully occupied an invaded area. For commercially valuable species, containment also depends on the traits of the species, the reason for its cultivation and the characteristics of the area where it is cultivated (Grice, 2006). Several methods are available to contain commercial and non-commercial species (Grice *et al.*, 2013). However, the methods need to be adapted to the dispersal capacity of the species and containment of each infestation or population may have to be attempted separately. Most importantly, containment may be treated only as a short-term measure, while other management methods are being developed for implementation.

The economic viability of "slowing the spread" was demonstrated for the invasive alien pest *Lymantria dispar* (gypsy moth) in North America (Sharov & Liebhold, 1998). In the forestry sector, the successful containment of gypsy moth was reported from the United States (from Wisconsin to North Carolina) where pheromone traps were used to disrupt mating of the moth or alternatively treating the population with *Bacillus thuringiensis* (Sharov *et al.*, 2002).

Similarly, in agriculture, sterile insect techniques may be used to contain invasive alien pests (**section 5.4.3.2**). Containment is a viable strategy when used within zoological or botanical gardens or when predator free fences are used to exclude invasive alien vertebrates from invading native wildlife reserves (Ringma *et al.*, 2018). Use of this method in marine ecosystems may be ineffective in the long-term but has been used as a rapid response plan to manage diseases in aquaculture in disconnected water systems. In 1997, *Styela clava* (Asian tunicate) was first noted invading an aqua-cultured *Mytilus edulis* (common blue mussel) in Prince Edward Island, Canada (Locke *et al.*, 2009). After confirming the identity of the species, a group of stakeholders implemented a containment strategy in 2001 to manage the species (Locke *et al.*, 2009). Transfer and harvest of blue mussels were restricted in tunicate-infested areas and responsible practices were encouraged. Although no cost-benefit-risk analysis was done, the results proved that benefits outweighed the costs. The manual handling and disposal costs totalled 0.24 Canadian dollar per kilogram of harvested mussel (Locke *et al.*, 2009). This forms a good example of a successful containment programme but was effective only in the short term. The use of a combination of methods (including chemical control) and long-term monitoring may be necessary to mitigate tunicate impacts and develop a sustainable mussel aquaculture industry (ACRDP, 2010).

5.5.5 Control

Successful invasive alien species control is generally assessed as the levels of invasive alien species suppression. Objective-driven invasive alien species management may also measure improvements to biodiversity and ecosystem services in the context of sustained ecosystem restoration **(Box 5.19)**. Invasive alien species control requires long-term monitoring for continued management actions so as to ensure sustained control. Long-term monitoring is also essential to assess efficacy and outcomes of management actions, and assess return on investments and benefits to local communities.

Lissachatina fulica (giant African land snail), native to East Africa, is listed as one among 100 of the world's worst invasive alien species (Lowe *et al.*, 2000). Recorded from over 50 countries in all continents except Antarctica, it causes

Box 5 19 **The Working for Water programme: Social benefits from controlling invasive alien plants.**

Control of widespread invasive alien species requires large-scale and continuous efforts to reduce their density. South Africa's Working for Water programme, introduced in 1995, took advantage of the need to clear invasive alien vegetation as part of a water conservation campaign and poverty relief programme by creating job opportunities for thousands of local people (e.g., 20,000 jobs per year over the first 15 years of the programme; Lukey & Hall, 2020; van Wilgen *et al.*, 2012). The programme also provided training in entrepreneurial and management skills and a sense of community among workers, especially women (Binns *et al.*, 2001). The programme addressed a national imperative to improve good quality of life of predominantly poor rural communities, while managing the spread of many invasive alien plants, and for some species, reducing the area of invasion (Wilson *et al.*, 2013). Although sustainability of the programme has been a concern (Binns *et al.*, 2001), the Working for Water programme has been ongoing for more than 25 years and is seen as a successful example of invasive alien species control which has brought ecological and social benefits in partnership with various stakeholders (Lukey & Hall, 2020). The programme contributed primarily to employment generation, rural development and water security.

significant impacts on crops (Sankaran, 2008). Physical and chemical methods were unsuccessful in managing the snail. Common molluscicides are useful for short term control but resulted in soil and water pollution and affected non-target snails. The Kerala Forest Research Institute, India has since developed a non-polluting method, which is effective for *Lissachatina fulica* management in the longer term. The control method involves two stages 1) baiting and 2) point chemical treatment, resulting in 100 per cent mortality without non-target impacts. Dead snails are buried in soil. Local communities have accepted this method and are now practicing it (Maneetha *et al.*, 2017).

Box 5 20 African smallholder farmer management of the recent *Spodoptera frugiperda* (fall armyworm) invasion.

Fall armyworm management adopted by African farmers uses combinations of chemical, physical, cultural or traditional methods (**Figure 5.24**; FAO, 2018b; Kansiime *et al.*, 2019; Tambo *et al.*, 2019; Murray *et al.*, 2019; Rwomushana *et al.*, 2018; Asare-Nuamah, 2020; Koffi *et al.*, 2020; Gebreziher *et al.*, 2021; Houngbo *et al.*, 2020; Tambo, Day, *et al.*, 2020; Tambo, Kansiime, *et al.*, 2020; Bariw *et al.*, 2020).

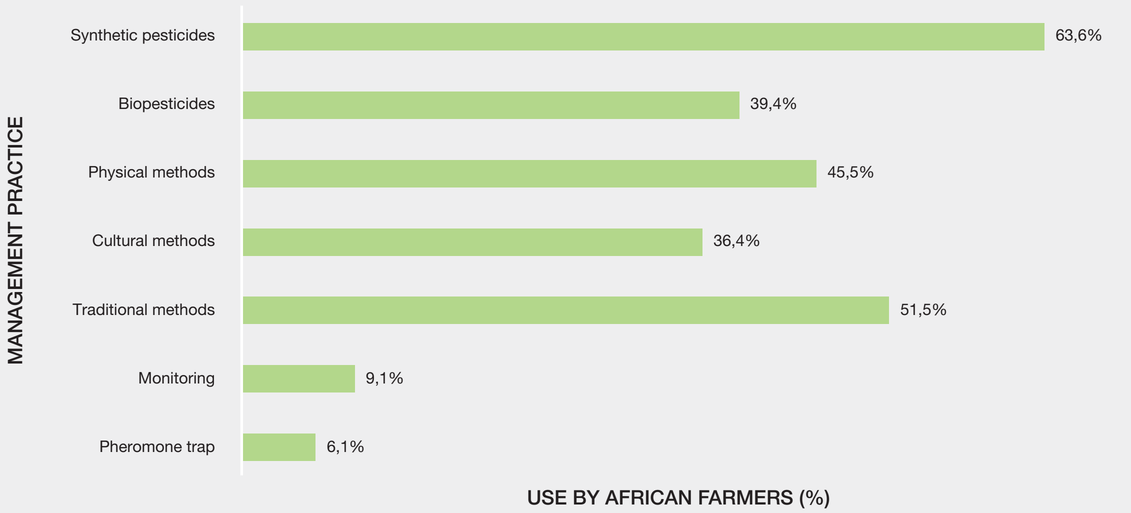


Figure 5 24 **Fall armyworm management adopted by African farmers uses combinations of chemical, physical, cultural or traditional methods.**

(FAO, 2018b; Kansiime *et al.*, 2019; Tambo *et al.*, 2019; Murray *et al.*, 2019; Rwomushana *et al.*, 2018; Asare-Nuamah, 2020; Koffi *et al.*, 2020; Gebreziher *et al.*, 2021; Houngbo *et al.*, 2020; Tambo, Day, *et al.*, 2020; Tambo, Kansiime, *et al.*, 2020; Bariw *et al.*, 2020).

Synthetic pesticides were the most commonly used control method (64 per cent of studies). Cultural methods (36 per cent of studies) involved agronomic practices such as early planting, intercropping with non-host plants, weeding of the field constantly to remove alternative host plants, push-pull technology and fertilization to produce healthy plants that are resilient to attack. Traditional methods (52 per cent of studies) included the application of household detergents, soaps, ash, sand or urea on larvae. Integrated pest management consisted of regular monitoring of maize fields for fall armyworm with the use of pheromone traps to monitor or capture the adults.

As fall armyworm was a new pest, farmers needed information on identification, biology, monitoring and effective control, including information on pesticide use and safety, but this was largely unavailable or inadequate (Nyangau *et al.*, 2020; Girsang *et al.*, 2020; Murray *et al.*, 2021; Tambo *et al.*, 2021). Pesticide cost was high and supplies and resources were low (Bariw *et al.*, 2020). Handpicking of larvae was labour-intensive (Chimweta *et al.*, 2020).

In terms of effectiveness, pesticides were most effective (Rwomushana *et al.*, 2018), while early planting, handpicking, planting resistant varieties, crop rotation and replanting were all perceived as highly to moderately effective in Namibia. Early planting and handpicking were considered relatively ineffective by farmers in Benin (Houngbo *et al.*, 2020). Ash application was considered ineffective in Namibia (FAO, 2018b) and Benin (Houngbo *et al.*, 2020). Further information can be found in **Supplementary material 5.11**.

China has lost millions of native pine trees (*Pinus tabuliformis* (Chinese pine) and *Pinus bungeana* (lace bark pine)) to *Dendroctonus valens* (red turpentine beetle) introduced from North America in logs in the 1980s, which has led to loss of tree cover leading to ecosystem change, lost carbon sequestration and biodiversity. To manage this invasive alien species, China has adopted an adaptive integrated management approach built on strong regulatory controls on timber movement and silvicultural, insecticidal and semiochemical trapping. The programme has limited the rapid pest spread and further impact on the native pine trees (Yan *et al.*, 2005; J. Sun *et al.*, 2013; Wan *et al.*, 2017).

Most Indigenous Peoples and local communities control invasive alien species through physical removal, especially invasive alien plants. Managing crop weeds on smallholder cropping lands in Africa is largely done by women and children, and is often their most time-consuming activity (Chikoye *et al.*, 2006; Orr *et al.*, 2002; Terefe *et al.*, 2020; Vissoh *et al.*, 2004); for example, *Opuntia* spp. (pricklypear) in East Africa (R. T. Shackleton *et al.*, 2017) as repeat weeding is required. For *Pontederia crassipes* (water hyacinth), physical removal has proved futile as the plant quicky grows back and seeds, which remain viable for 15 years, can spread through animal faeces (Heuzé *et al.*, 2015; Gopal *et al.*, 2019). Indigenous Peoples and local communities from western Kenya uproot the parasitic *Striga hermonthica* (witchweed) from maize plantations but control is ineffective (Oswald, 2005). Ineffective management strategies can also have social impacts. Pastoralists of Baadu (Ethiopia) failed in the efforts to remove *Prosopis juliflora* (mesquite), and this has led to changes in the good quality of life including social conflicts (Rettberg, 2010; Rettberg & Müller-Mahn, 2012).

Indigenous Peoples and local communities have attempted multiple methods to control *Spodoptera frugiperda* (fall armyworm) as it spread across Africa and Asia from the Americas to save their livelihoods, but often to little effect. In Ghana, some Indigenous Akan People applied So Klin (a washing detergent solution) to reduce the negative impacts of fall armyworm (Asare-Nuamah, 2020; **Box 5.20**; **Supplementary material 5.11**). More traditional approaches included cultural and spiritual practices and management by fire. The Yellomundee Aboriginal Bushcare in Australia (Barber & Glass, 2015) believe that it is "a cool fire that burns the invasive alien plants but allows native species to regenerate". Early season patch fire management removes biomass and stimulates native seedlings while not burning surrounding trees. This traditional approach to fire management is now widely recognized, supported and practiced across Northern Australia. In the Kimberley, Western Australia, the place-based (a type of site-based) invasive alien plant management approach, developed by rangers on behalf of the Bunuba People, protects sacred sites (Aiken *et al.*, 2015). The Rajbanshi People from North Bengal in India practices sacred bathing in winter and autumn, such as Maghali sinan and Bauni sinan and also worships rivers at the onset of monsoon season to get timely rains, which will help the fight against invasive alien invertebrates (A. D. Gupta, 2015). Local farmers in Lao People's Democratic Republic use wood ashes for coating stems of crops to protect them from invasive alien invertebrates (Upadhyay *et al.*, 2020). A community-led approach often results in success. The Holok system of customary law and other cultural practices of the Ifugao people of Hingyon utilizes parts of more than 25 plants to produce a biopesticide against several invasive alien invertebrates. The holok, as traditionally practiced, was part of the hongan di pageh, the system of Ifugao rituals on rice culture (IPBES, 2020).

5.5.5.1 Mechanical, manual and chemical methods

See **section 5.4.3.2** for more information.

5.5.5.2 Lethal control programmes

There is a very high success rate of invasive alien vertebrate eradication programmes on islands (**section 5.5.3**), and there are some successful programmes removing mammals from within defined larger land mass boundaries (Robertson *et al.*, 2017). The vast majority mainland landscape management programmes to suppress uncontained invasive alien vertebrate based on culling (lethal population suppression) have however been ineffective (reviewed by Hone, 2007; **section 5.4.3.2d**). This is because lethal control programmes are generally poorly planned and implemented based on:

- inadequate understanding of population sizes, distributions and metapopulation dynamics of the target in time and space,
- ineffective tracking of populations in hunting programmes leading to culling mainly being concentrated where and in seasons when the target is most abundant (minimizing the chance of suppressing a population below an ecological impact threshold), limiting effectiveness of removal with respect to environmental impacts;
- lack of sustained investment and activity leading to only temporary population suppression; and
- failure to use as part of integrated management including fencing to protect cleared areas and sustain the short-term management benefits.

Developing effective selective baits and trapping is also a strong criterion for success, particularly for shy and hard to track feral animals such as cats. Public opinion is also likely to affect management programme success. In the United States, advocates for feral *Felis catus* (cat), listed as one of the 100 worst invasive alien species, blocked federal legislation that would have funded removal of various invasive alien species, potentially including cats, from national wildlife refuges (Longcore *et al.*, 2009). In Italy, management of invasive *Sciurus carolinensis* (grey squirrel) was hindered by a lack of public acceptance (Hulme, 2006). Although most non-government organizations supported management using humane euthanasia of the squirrel, strong opposition from animal rights organizations interrupted the activities allowing subsequent squirrel range expansion (Genovesi & Bertolino, 2001). Effective invasive alien predator management has led to increased abundance of native animals in Australia (Bengsen *et al.*, 2012; Doherty *et al.*, 2017) and New Zealand (O'Donnell *et al.*, 1996; PREDATOR Free NZ, 2021).

5.5.5.3 Classical biological control programmes: successes and failures

The practice of classical biological control to suppress populations of invasive alien species has a successful history of well over 100 years (**section 5.4.3.2f**). Biological control is a widely used invasive alien species management approach in many countries and continues to be applied to manage a range of invasive alien plants, invertebrates and to a lesser extent plant microbes and a few invasive alien vertebrates (Cock *et al.*, 2016).

For invertebrate targets, the BIOCAT database shows that there have been 6158 biocontrol agents released to control invasive alien invertebrates before 2010, and the probability of successful establishment and impact of introductions continues to improve (Cock *et al.*, 2016). This led to the successful management of 172 different target organisms. Van Driesche *et al.* (2010) reviewed releases against environmental targets and found a 62 per cent success rate for complete control with a further 19 per cent partially controlled. *Oryctes rhinoceros* (coconut rhinoceros beetle) is a major pest across the Pacific islands that has been widely managed using a well-established classical biocontrol agent, *Oryctes rhinoceros* nudivirus (OrNV), for many years, however recently beetle numbers have been rapidly increasing, severely disrupting nature's contributions to Indigenous Peoples and local communities through free access to coconuts across many islands in the Pacific. This resulted in Vanuatu declaring a national emergency. Recent research suggests the effectiveness of the virus has declined and this may be a rare example where the invasive alien species target has generated resistance to the biocontrol agent (Etebari *et al.*, 2021).

The global catalogue of biocontrol agents and their use against target alien invasive plants shows that up to 2018, 468 biocontrol agents have been released against 175 species of invasive alien plants across 48 families and 90 countries. Some form of successful control was achieved against 65.7 per cent of the plant species targeted, for which sufficient time has elapsed to assess effectiveness. One third of targets no longer required any other form of control (Schwarzländer *et al.*, 2018; e.g., **Box 5.21**). The biological control programme against *Ambrosia artemisiifolia* (common ragweed) in China, the pollen of which has a very high allergy rate in humans leading to high medical costs, has released two biological control agents (*Ophraella communa* (ragweed leaf beetle) and *Epiblema strenuana* (ragweed borer)). These biocontrol agents successfully suppressed the target in southern China, however in colder northern China biological control needs to be supplemented by chemical control and restoration with native plants (Wan *et al.*, 2017). Biological control effectiveness is related to the level of abundance of the target plant in the native range, the mode of reproduction (sexual *versus* asexual) and the habitat type (aquatic *versus* terrestrial; Paynter *et al.*, 2012). Although many biological control programmes targeting invasive alien species take many years with no guarantee of success, this approach remains very cost-effective because the control benefits, when they occur, are generally high and self-sustaining (Briese, 2000). For invasive alien plants,

Box 5 21 **Case study of biological control of *Mikania micrantha* (bitter vine) in the Asia-Pacific region.**

Mikania micrantha is a fast-growing invasive alien plant native to Central and South America. It invades plantations and agricultural systems, thereby reducing productivity threatening the livelihood of rural communities in the Asia-Pacific region (Anitha *et al.*, 2017; Day *et al.*, 2016; Ellison & Sankaran, 2017). A microcyclic rust fungus *(Puccinia spegazzinii)*, which causes necrosis of leaves and cankers on the stem and petioles in the native range of the species, was introduced into India in 2006 and then in China, Papua New Guinea, Fiji, Guam, Palau, Vanuatu and the Cook Islands (2006 – 2012) (Day, Kawi, Fidelis, *et al.*, 2011; Day *et al.*, 2016; Orapa, 2017). The rust established in five countries (Taiwan, Province of China, Papua New Guinea, Fiji, Vanuatu and the Cook Islands) and has kept the spread of the bitter vine well under control, especially in Papua New Guinea and Vanuatu (Ellison & Cock, 2017). However, in India, the rust fungus failed to survive in the field apparently due to a low inoculum load and inappropriate time of release (Sankaran & Suresh, 2013). Paucity of resources prevented further releases in India.

a third of programmes only release one biocontrol agent and often one agent provides the necessary control, but as selecting agents based on likely future effectiveness is hard, the release of multiple agents is often required (Schwarzländer *et al.*, 2018). When such programmes are unsuccessful, termination is generally more to do with perceived levels of risk to non-target native species, failed agent establishment or lack of funding and political will than that all biocontrol agent options have been exhausted (Fowler, 2000; Sankaran & Suresh, 2013).

Approximately 50 per cent of classical biological control programmes for invasive alien plant or invertebrate species do not deliver much effective return on investment (Cooke *et al.*, 2013; Julien *et al.*, 2012; Waterhouse & Sands, 2001). The benefits of successful programmes, however, can more than pay for projects that were not successful. In Australia, where a total benefit-cost assessment has been undertaken for classical biological control against invasive plants in agricultural systems, the national effort over 100 years gave a return on investment of 23:1 including the costs of both successful and unsuccessful programmes. This was an annual benefit of 95.3 million Australian dollar (AU$) a year in 2006 (Page & Lacey, 2006). As the monetary benefits cannot easily be measured for the impacts of invasive alien plant targets in natural ecosystems, based on the number and benefit magnitudes of successful programmes, the returns on investment were considered at least equivalent against invasive plants. Benefit-cost ratios of six programmes in South Africa ranged from 34:1 to 4333:1 (van Wilgen *et al.*, 2004). Some invasive alien plant species are best managed by integrating biological control with other management practices (Moran *et al.*, 2005). Evidence indicates that biological control alone may not be efficient to manage some of the invasive alien plants where integrated management is the most viable option.

A viral-based classical biological control programme against *Oryctolagus cuniculus* (rabbits) in Australia has also been highly successful and also had the support of the local peak body on the prevention of cruelty to animals (RSPCA Australia; **section 5.4.3.2f**). Since the release of the first biological control agent in the late 1940s the programme had delivered AU$ 70 billions of benefit by 2011 (Cooke *et al.*, 2013; **Supplementary material 5.12**). Classical biological control has been considered but not adopted against other invasive mammals, invasive alien fish, amphibians, reptiles and birds (CBD, 2019; A. W. Sheppard *et al.*, 2019). Marine invasive alien species have not been targeted for biological control although the approach has been considered (Simberloff, 2021). Secord (2003) and Lafferty and Kuris (1996) have undertaken reviews of the opportunities and the risks and doubt its relevance.

The application of rigorous and internationally agreed risk analyses starting in the 1950s has reduced incidents of unpredicted non-target impacts to a very low and largely predictable level, a trend that may continue with the systematic inclusion of molecular tools, behavioural studies, chemical ecology and future scientific and analytical advancements (**Chapter 3**, **section 3.3.5.2**). There are exceptions, such as *Harmonia axyridis* (harlequin ladybird) in Europe (Brown *et al.*, 2008; **section 5.4.3.2** for other non-target impacts). Direct non-target impacts from biological control programmes have been repeatedly reviewed and found to be predictable and minor compared to the native ecosystems' benefits from control, except for some early unregulated releases of generalist predators (e.g., the release of cats and mongoose on islands). Indirect impacts have received much less attention being less obvious and more difficult to measure. Where studied they are minor and ephemeral if control is achieved and generally confined to areas in close proximity to the target invasive alien plant for biocontrol agents that have undergone rigorous risk assessment. The completely unregulated introduction of *Tyto alba* (barn owl) in Hawaii in the late 1950s to control rats is a rare but unsurprising counter example, although this release did not follow the precautionary approach now applied in the context of modern classical biological control programmes. By 1966, the owls were established and breeding and a recent review found these owls were an important avian predator of at least eight seabird species (Raine *et al.*, 2019). A management programme to control owl populations has been undertaken in 2015. Biological control in any form, like most other management tools, is not risk-free (CBD, 2018).

5.5.6 Management in an ecosystem restoration context

Restoration of an ecosystem after invasive alien species control is both expensive and hard to achieve, unless the ecosystem retains a strong regeneration potential. This is especially true in marine ecosystems where invasive alien species management has proven to be largely ineffective (Lopez *et al.*, 2006). Integrating management and restoration into an adaptive management approach requires long-term monitoring to assess efficacy, outcomes and timely detection of lost resilience and reinvasion. Benefits of management, particularly to local communities, also need to be evaluated. In successful cases of restoration in terrestrial ecosystems, efforts are limited in space and time and goals are clearly defined and achievable with available resources (IPBES, 2018). See **section 5.4.3.3** for a description of site-based integrated invasive alien species management with ecosystem restoration strategies. China has been attempting an ecosystem restoration project for controlling *Sporobolus alterniflorus* (smooth cordgrass) introduced in 1979, which now covers hundreds of thousands of hectares in the Yangtze River estuary. The Shanghai government is spending 1.3 billion Yuan to control *Sporobolus alterniflorus* invasion and restore habitats for migratory birds (Wan *et al.*, 2017). The integrated management includes cofferdam construction for containment, mechanical harvesting, flooding, revegetating with native plants and managing water levels (Xiao *et al.*, 2011).

In a review on site restoration as a part of controlling invasive alien species, Kettenring and Adams (2011) observed that, a) the use of herbicides effectively but temporarily controlled invasive alien plants but did not lead to significant native revegetation; b) prescribed fire reduced the biomass of native species and increased the biomass of the invasive alien species; and c) cutting/removal of the invasive alien species slightly decreased invasive alien species biomass but not that of native species. However, most studies failed to quantify the effectiveness of ecosystem restoration since they had failed to measure the initial status of native vegetation. This has led to inconsistent conclusions regarding the best invasive alien plant control option that may lead to the most effective ecosystem restoration.

One of the common methods to restore terrestrial ecosystems invaded by invasive alien plants is to plant fast-growing native (annual/perennial) species or disperse seeds of such species following effective management of the invasive alien species. Though such ecosystem restoration attempts may not be sufficiently efficient to enhance resistance to invasive alien species, growth and spread of planted native species may help to suppress regeneration of the invasive alien species community by filling recruitment niches (Byun *et al.*, 2013; Byun, Oh, *et al.*, 2020). However, large seedbanks of the invasive alien species may often interfere with these attempts. Therefore, success of ecosystem restoration depends on ensuring a well-established seed bank of native plants at the site and on long-term monitoring of the restored habitats to ensure establishment of the planted seedlings and to manage re-invasions (Byun, de Blois, *et al.*, 2020; Byun *et al.*, 2018). Assisted natural regeneration of native plants by protecting the area from grazing, fire and other interventions may also help successful ecosystem restoration. Local community cooperation is essential for the success of assisted regeneration.

Field experiments have shown that a good knowledge of the functional-trait-based biotic resistance and diversity-resistance in the community will help to achieve successful restoration of native communities on sites where invasive alien plants were successfully managed. Resistance to invasive alien species may be associated with community functional diversity (Byun, Blois, *et al.*, 2020; Byun *et al.*, 2013, 2018; **Chapter 1**, **section 1.4.3**), and functional diversity could be (based on trait complementarity) a good indicator of invasibility. A recent study of communities invaded by *Phragmites australis* (common reed) in Canada (Byun, de Blois, *et al.*, 2020; Byun *et al.*, 2013) proved that functional diversity-based resistance to invasive alien species differs between invasive alien species, and restoring functional diversity could provide resistance against multiple invasive alien species. It is certainly prudent to restore functional diversity as part of ecosystem restoration since the process of restoration will be easier if functional diversity is not lost.

5.5.7 Management costs

The global economic cost of invasive alien species is over \$1 trillion and the cost is rising (**Chapter 4**, **Box 4.13**; (Diagne *et al.*, 2021). This cost represents documented expenditures with management of biological invasions (e.g., prevention, control and monitoring) and economic losses associated with the impact of invasive alien species. The global reported costs of management of biological invasions (excluding impacts of invasive alien species) totalled \$120.5 billion (2017 US\$ values) over the last 50 years (**Figure 5.25**; (Diagne *et al.*, 2020). The geographic distribution of management costs **(Figure 5.26)** shows that the documented costs were highest in the Americas (\$103.5 billion), followed by Asia-Pacific (\$6 billion) and Africa (\$5 billion). Management costs for invasive invertebrates were the highest (\$29 billion), followed by plants (\$5.7 billion) and the management costs were highest for terrestrial ecosystems (\$107.8 billion). Data on whether higher management costs were spent on prevention *versus* management were equivocal, but funds being spent globally on research for the management of biological invasions were low (\$2.78 billion). On a global scale, a study showed that eradication of invasive alien species can make substantial savings on costs devoted to the protection of threatened native species (Jones *et al.*, 2016), suggesting that eradication of invasive alien species is a very cost-effective investment for protecting threatened and endangered species in comparison.

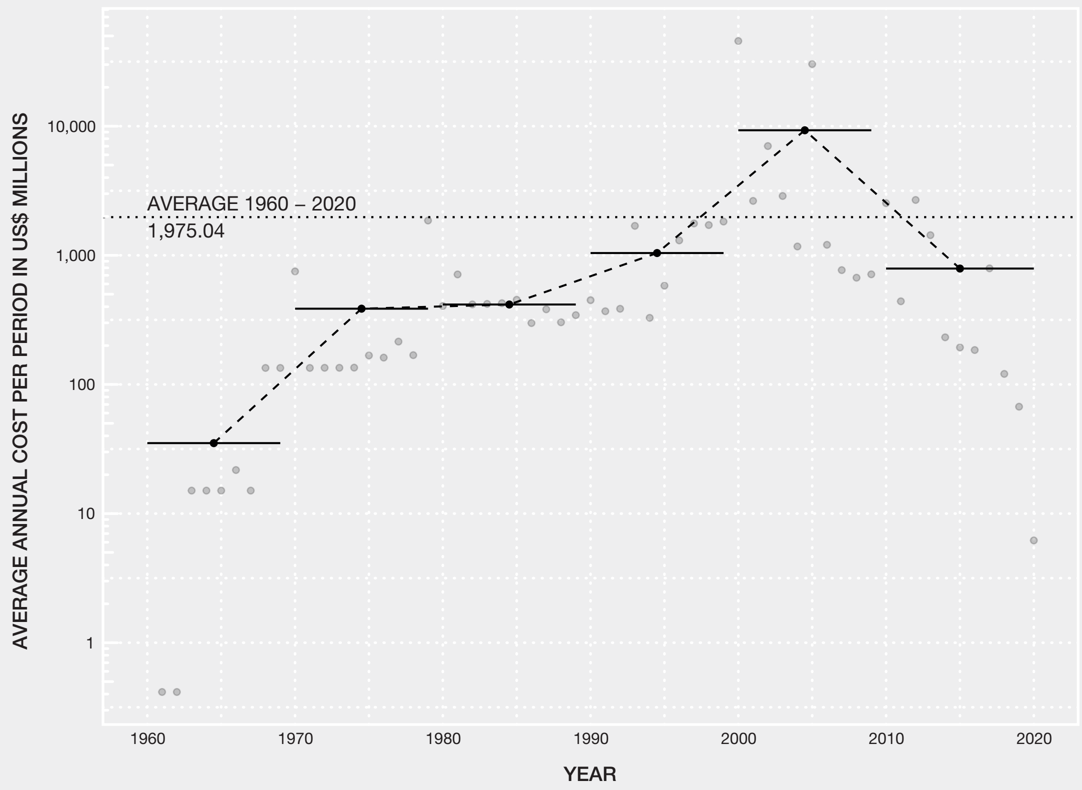


Figure 5 25 **Annual cost reported globally with the management of biological invasions between 1960 and 2020 (2017 monetary values).**

Light dots represent annual average cost reports and dark dots (with lines on each side) connected by dashed lines represent the decade averages. Data source: Diagne *et al.* (2020).

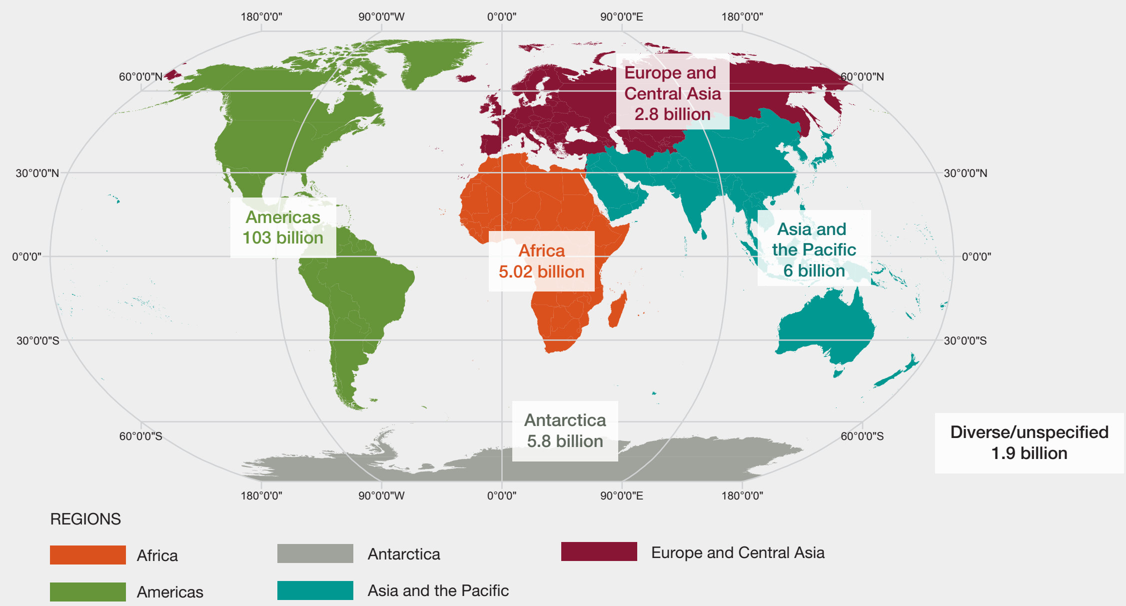


Figure 5 26 **Total reported economic cost of management of biological invasions actions across all IPBES regions (in USD) between 1960 and 2020 (2017 monetary values).**

Data source: Diagne *et al.* (2020).

5.6 CONTEXT SPECIFIC CHALLENGES AND KNOWLEDGE GAPS IN MANAGEMENT

5.6.1 Context specific challenges

5.6.1.1 Challenges to management success across taxa and ecosystems

The conceptualized invasion management-invasion continuum **(Figure 5.1)** provides a simplified schematic of the typical process and potential management options for biological invasions. However, the context (where in the invasion continuum), invasive alien species type (Booy *et al.*, 2020), unique environmental conditions (A. W. Sheppard *et al.*, 2002) and the economic costs for each management scenario will differ between situations (Pluess, Jarošík, *et al.*, 2012). For example, an analysis of 173 eradication attempts across 94 invasive alien species showed that eradication was more likely in anthropogenic than in semi-(natural) sites (Pluess, Jarošík, *et al.*, 2012; **section 5.5.3**). The study also showed that eradication attempts are only likely to be successful if initiated within four years after introduction (Pluess, Jarošík, *et al.*, 2012). Globally it has been shown that alien vertebrates are easier to eradicate than alien generalist invertebrates, pathogens and plants (Booy *et al.*, 2020). Plants and fungi, for example, produce seeds/spores or other propagules which are hard to find and may remain dormant for many years (Mack & Lonsdale, 2001). One classical example of eradication is that of *Myocastor coypus* (coypu) from a large region of south-eastern England (Gosling & Baker, 1989). Although highly context- and scale-dependent, there are examples of aquatic plants and freshwater fish being eradicated (Simberloff, 2021). In Norway, the invasive alien fish that have been successfully eradicated include *Phoxinus phoxinus* (European minnow), *Rutilus rutilus* (roach), *Esox lucius* (pike) and *Coregonus lavaretus* (common whitefish) (Bardal, 2019). The feasibility of eradication of invasive alien species in marine ecosystems at any scale is, however, generally small (Booy *et al.*, 2020).

For individual invasive alien species, different populations within one habitat will vary in their density and impacts (**section 5.3**; Dassonville *et al.*, 2008). This variability alters options for optimal management. For example, *Undaria pinnatifida* (Asian kelp) has invaded most temperate regions worldwide, but conditions for successful biological invasion could not be generalized across regions (Epstein & Smale, 2017). Also, in the marine context, invasive alien species are notoriously difficult to control, because the whole system is open and there are complexities in detection and implementing and evaluating responses to management actions (Simberloff, 2021).

An analysis of 76 case studies documenting the management of invasive alien species by Indigenous Peoples and local communities showed that plants are the most frequently reported target of management (**Supplementary material 5.1**), although plants are relatively difficult to eradicate. Vertebrate animals are also often targeted by Indigenous Peoples and local communities for management. However, the attempts of managing invasive alien animals, especially mammals, on Indigenous lands may not be successful, because of cultural or spiritual conflicts rather than the biological characteristics of the taxa (Koichi *et al.*, 2012; Peltzer *et al.*, 2019). There are few case studies reporting the management of invertebrates, fungi and pathogens implemented by Indigenous Peoples and local communities. The majority of the case studies reviewed have focused on terrestrial ecosystems, whereas there are much fewer studies documenting the attempts in freshwater and marine ecosystems (**Supplementary material 5.1**). This might imply that Indigenous Peoples and local communities have not actively attempted management of invasive alien species in aquatic ecosystems since it is notoriously less feasible than terrestrial ecosystems (but see **section 5.3.1.2** for examples of uses of aquatic invasive alien species by Indigenous Peoples and local communities leading to the control of species).

Management of cryptic, marine and infectious and zoonotic diseases remain a challenge. However, recent advances in environmental DNA are improving detection capability, and the improvements in automated underwater vehicles are making detection and management of marine invasive alien species easier, but authentic identification of marine species is one of the greatest obstacles (**sections 5.4.4**, **5.5**). For zoonotic (e.g., Severe Acute Respiratory Syndrome Coronavirus 2 (SARS-CoV-2), Covid-19 (C. R. Wells *et al.*, 2020)) and other infectious diseases (e.g., Foot and mouth disease; (Tildesley *et al.*, 2006), management is also benefiting from genetic surveillance tools as prevention and preparedness are critical to avoid impacts.

5.6.1.2 Management challenges for conflict species

The concept of an invasive alien species is a human construct and therefore perceived risks, benefits, costs and impacts from them vary depending on a diversity of human perspectives (Kelsch *et al.*, 2020; **Chapter 1**, **section 1.5.2**; **Chapter 4**, **section 4.1.2**; **Box 4.9**). There are many cases where species provoke strong disagreement on the requirement of management between stakeholders. The most common conflict comes from species that offer both benefits in some sectors (e.g., agriculture and nature's contributions to people) and negative costs or impacts in other sectors (e.g., biodiversity, nature's contributions to people and good quality of life) (Pejchar & Mooney, 2009). This value-based context dependency of particular species is often significant enough to prevent effective decision-making and management (R. Gregory *et al.*, 2006; Kelsch *et al.*, 2020; **Table 5.9**). Public value put on common pets such as *Felis catus* (cat), *Oryctolagus cuniculus* (rabbits) or gold fish often biases understanding of their invasive impacts (Nogales *et al.*, 2013). Attraction to charismatic species, such as *Myiopsitta monachus* (monk parakeet; (Crowley *et al.*, 2019) and hedgehogs (C. Jones *et al.*, 2005) can also bias recognition of environmental impacts. In such cases effective community-based communication programmes generally help (Jarić *et al.*, 2020).

Conflicts based on value systems include utilitarian, moralistic, spiritual, humanistic, naturalistic/dominionistic/aesthetic and risk perceptions (Estévez *et al.*, 2015). They are based on people's different beliefs, knowledge and experience with the invasive alien species and the social, cultural, landscape and policy contexts (Shackleton, Adriaens, *et al.*, 2019). Perceptions and values differ between countries and regions, policymakers, communities, managers, conservationists and Indigenous Peoples and local communities (Kelsch *et al.*, 2020), even if globally, values regarding environmental conservation are aligning. Some species may have been deliberately introduced for a particular service but, while providing that service, have negative impacts on other sectors. For example, introduced *Sus scrofa* (feral pig) are culturally important in Hawaii and are hunted for subsistence, ceremony and recreation (Pejchar & Mooney, 2009), but are considered keystone species in driving and maintaining alien plant invasions and causing forest ecosystem disruption, and therefore are primary modifiers of the remaining Hawaiian rainforest (Loope *et al.*, 2013). Elsewhere in the United States, introduced *Sus scrofa* are valued by hunters but cause annual losses to six crops of nearly $200 million, and potentially carry zoonoses or diseases that may affect wildlife and livestock (Lewis *et al.*, 2019). Indigenous Peoples and local communities often value invasive alien species culturally if they have become a food or livelihood source, and therefore may have adopted or adapted to use and value a particular invasive alien species that has established in their communities, even if the same species is causing significant environmental impacts (e.g., cats in Australia (I. Abbott, 2002) or mesquite in Africa and India (Chandrasekaran & Swamy, 2016; Mbaabu *et al.*, 2019) for which compromises can be found). For example, for feral ungulates, one solution adopted by Indigenous Peoples and local communities in Australia is the selection and fencing of a priority area (e.g., Heritage sites and high biodiversity wetlands) with the animals removed from these sites but still harvested sustainably outside the protected zone (E. Ens *et al.*, 2016). A similar solution was found in a hunting programme in Argentina **(Box 5.6)**. Many other invasive alien species are also used for sport. Stocks of *Micropterus salmoides* (largemouth bass) are supplemented for recreational fishing in South Africa despite severe ecological impacts on macroinvertebrate fauna and communities (O. L. F. Weyl *et al.*, 2015; P. S. R. Weyl *et al.*, 2010). Trouts are a globally prized recreational fish, restocked into rivers every year in many countries outside their native range, however, this conflict has been analysed only in South Africa (Marire, 2015).

Participation of different stakeholder communities in invasive alien species management can also lead to conflicts and perverse outcomes. Harvesting and recreational hunting are often considered as incentives for invasive alien vertebrate management, but whether their inclusion as part of management programmes improves or simply perpetuates the problem as the target becomes a resource, is strongly debated (C. Booth, 2010; Nuñez *et al.*, 2012; Gentle & Pople, 2013; Zivin *et al.*, 2000). Australia is the world's largest exporter of goat meat (Meat & Livestock Australia, 2021) based on harvesting feral goats as an economic safety net for rural communities in times of drought. This is a significant impediment to widespread management of the environmental impacts of feral goats (D. M. Forsyth *et al.*, 2009).

In the horticultural sector, about 75 per cent of the global established alien flora are grown in domestic gardens (van Kleunen *et al.*, 2018), which presents a particular challenge when seeking to gain public support for their management. Generally, when supported by information campaigns and when the impacts after escaping the garden fence are realized, the public understands the need to avoid some plant species in their gardens and nurseries voluntarily agree not to stock them (Burt *et al.*, 2007). Another approach is to encourage the benefits of gardens with native plants (Shaw *et al.*, 2017). Codes of conduct and agreed prohibited lists have been developed between government agencies responsible for managing invasive alien plants and the nursery trade to address these conflicts (Atkinson & Sheppard, 2000). These include the horticultural industry, conservation and environmental agencies and the plant protection sectors (including regulatory bodies) and can improve management of invasive alien plants without impacting on an important economic industry (Heywood & Brunel, 2011).

Some alien plant species with high economic value become invasive in other contexts, for example for forage production or plantation forestry (van Wilgen & Richardson, 2014). Scasta *et al.* (2015) documented that the cultivation of *Lespedeza cuneata* (sericea lespedeza), a declared invasive alien species by the state invasive council, for forage production was publicly recommended in the state of Alabama. Similarly, Nuñez *et al.* (2017) documented a case where forestry plantation of a pine species, *Pinus contorta* (lodgepole pine), had been heavily subsidized by national government until recently, even though high invasiveness of the species has been recognized. Agreeing on management of some of the agriculturally valuable but environmentally harmful invasive alien grasses (e.g., African grasses in Australia; Cook & Dias, 2006) remains a challenge, but biological control of *Acacia mearnsii* (black wattle) timber trees in South Africa found a compromise by selecting biological control agents that only reduce propagule production (i.e., flower and seed feeding agents) (Impson *et al.*, 2011; Impson *et al.*, 2021). This illustrates how different stakeholder communities can be brought together to understand the different perspectives and seek a joint solution. Information campaigns for the general public around charismatic invasive alien species also play a key role in raising awareness in the community. Explicit consideration of the factors creating the different views through education and inclusion is critical if management measures are to be mutually agreed (Jarić *et al.*, 2020).

Table 5 9 **Examples of invasive alien species that were intentionally introduced for beneficial purposes, conflict resolution and potential management response (Chapter 4, section 4.6.4).**

Groups include terrestrial plant; freshwater/marine plant; microorganism; bird/fish/mammal/reptile; insect.

Invasive alien species	Taxonomic group	Time and location of introduction	Native range of introduced species	Primary purpose of the introduction	Co-operative efforts to develop management options
Lates niloticus (Nile perch)	Vertebrate (freshwater fish)	Lake Victoria; 1960s	Afrotropical; Congo, Nile, Senegal, Niger, Lake Chad, Volta, Lake Turkana	To promote the fisheries industry as the dominant endemic haplochromine species were perceived to have low economic value (Njiru *et al.*, 2005)	The Kenyan, Ugandan and Tanzanian governments established a regional mechanism in 1994 – Lake Victoria Fisheries Organization – to coordinate the management and conservation. The three countries agreed to enforce legislation and regulations to protect the lake and its basin (Njiru *et al.*, 2005).
Procambarus clarkii (red swamp crayfish)	Invertebrate (freshwater crustacean)	Present in 40 countries across all continents except Australia and Antarctica (Nunes *et al.*, 2017; Oficialdegui, Sánchez, *et al.*, 2020)	Southern United States and north-eastern Mexico	Aquaculture	In one example in Europe, where the red swamp crayfish has a high economic value, legislation regulating the red swamp crayfish on the basis of biodiversity protection was overridden to allow continued use due to public opposition and socioeconomic interests. Therefore, the legislation did not achieve the desired environmental outcomes, leading to the recommendations that context specific legislation is more likely to receive wider support (Oficialdegui, Delibes-Mateos, *et al.*, 2020).
Robinia pseudoacacia (black locust)	Terrestrial tree	Europe	North America	Wood and honey production, amelioration and soil stabilization (Vítková *et al.*, 2017)	Societal concern resulted in the species not being included in the list of regulated species at the European level. In some countries, management is based on site-specific approaches leading to tolerance in selected areas and strict eradication at sites of high conservation value.
Prosopis juliflora (mesquite)	Terrestrial tree	35 countries in Africa; over 20 countries in Asia and the Pacific	The Americas	Soil stabilization and to provide fuel and livestock fodder It was introduced into South India for fuelwood purposes and to benefit the dryland economy	In one region a national plan to manage the invasion is under development, driven by bottom-up concerns, as a community requested compensation from the government after losing their cattle due to the effects of *Prosopis juliflora*. As it was introduced in a government programme (Shackleton *et al.*, 2014), the community was awarded compensation (Castillo, 2019). The use of *Prosopis juliflora* is a socio-economic concern in southern India where management is a complex issue as charcoal from the tree is a source of income for local people (Walter & Armstrong, 2014). Increased use of the wood through proper silvicultural management was proposed to control spread.

Table 5 9

Invasive alien species	Taxonomic group	Time and location of introduction	Native range of introduced species	Primary purpose of the introduction	Co-operative efforts to develop management options
Grasses and legumes (8200 species; V. M. Adams & Setterfield, 2015; G. D. Cook & Dias, 2006)	Terrestrial plant	Australasia	All continents	For pastoral improvement	Of all the species introduced, twice as many became invasive alien species than became useful (Lonsdale, 1994). Management options are being developed for two species, *Andropogon gayanus* (tambuki grass; V. M. Adams & Setterfield, 2015) and *Cenchrus ciliaris* (buffel grass; Grice *et al.*, 2012).
33 *Acacia* spp. including *Acacia mearnsii* (black wattle; Magona *et al.*, 2018)	Terrestrial trees	South Africa	Australia	For timber and as ornamentals	Agreeing and selecting biological control agents that only reduce propagule production (i.e., flower and seed feeding agents; Impson *et al.*, 2011, 2021).
Bombus terrestris (bumble bee)	Invertebrate (Insect)	Japan	Africa, Asia and Europe	For pollination of commercially important crops (Inoue *et al.*, 2008)	In principle, introduction, breeding and release are prohibited by the Invasive Alien Species Act, but farmers may use bumble bees on the condition that measures to prevent escape be taken and official permission be obtained (Goka, 2010; Lohrmann *et al.*, 2022).
Capra hircus (goats)	Terrestrial mammal	Mexico, Guadalupe Island	Asia	Meat production	Goats were introduced in the early 19th century by fur traders to have fresh meat. Later, there were permits from Mexico's government to use the goats as dry meat. Overgrazing by goats decreased forest coverage from 3,850 hectares to 85 hectares, while some vegetation communities disappeared. Because of the latter, with the support of federal government agencies (including the Mexican Navy), the local fishing community and the specialized private organization Grupo de Ecología y Conservación de Islas, the goats were eradicated (Aguirre-Muñoz *et al.*, 2011). The eradication of goats took place between 2003 and 2006. Seedlings of endemic trees that were absent in 2003, and species of plants believed extinct, reappeared, including species not seen in 100 years. To date, the vegetation has recovered rapidly, both naturally and through active ecosystem restoration (Luna-Mendoza *et al.*, 2019).

5.6.1.3 Management strategies for biological invasions under climate change and changing land-use as multiple drivers of change

Climate change is a driver that facilitates biological invasions (**Chapter 3**, **section 3.3.4**), and associated extreme climate events increase ecosystem susceptibility to biological invasions (Diez *et al.*, 2012; **Chapter 3**, **section 3.4**). Climate change and habitat loss or conversion are linked. Climate change influences land-, freshwater- and sea-use, which adds to the susceptibility of ecosystems to invasive alien species (**Chapter 2**, **section 2.1**, **Chapter 3**, **section 3.5.1**). Invasive alien species reduce the resilience of ecological communities and habitats to extreme events (Godfree *et al.*, 2019), therefore, prevention and management can increase the long-term climate change functional resilience of threatened ecosystems and habitats. In short, climate change poses increasing challenges for the management of biological invasions (Walther *et al.*, 2009). The interactions between climate change, habitat change and invasive alien species will alter drivers that facilitate biological invasions, resulting in new pathways of introduction, vector efficacy and species previously environmentally constrained overcoming establishment, reproduction and spread barriers (**Figure 5.27**; Walther *et* *al.*, 2009; **Chapter 3**). While models and scenarios give insights into the trends of likely impacts of climate change on invasive alien species (**Chapter 1**, **section 1.6.7.3**; **Chapter 2**, **sections 2.6.2**, **2.6.3**, **2.6.4**), mainstreaming these concepts into action to minimize future impacts will be challenging (Hellmann *et al.*, 2008). Similarly, building concerns related to management of biological invasions into climate change response planning is also essential, since ecosystem resilience to climate change is eroded by invasive alien species. This imperative cuts across the many sectors involved in climate planning, including human health, agriculture and aquaculture, forestry, fisheries management and wildlife conservation; it is acutely essential when co-planning adaptive management with Indigenous Peoples and local communities (**Chapter 4**, **section 4.7.2**; **Chapter 6**, **sections 6.1.1**, **6.3.1**).

The individual or synergistic effects of increased carbon dioxide levels, changes in air and seawater temperature, floods and droughts, increased frequency and intensity of fire regimes, higher saltwater incursions, changes in ocean currents, extreme events and precipitation patterns, and their interactions with invasive alien species is likely to be highly uncertain (Walther *et al.*, 2009). Future management of biological invasions will need to adapt, based on knowledge of how potential risks and impacts will vary with changing climate drivers (e.g., spatio-temporal rainfall shifts; Beaury *et al.*, 2020). Current "sleeper" (i.e., invasive alien species of low apparent risk; Hulme, 2020b) may become more invasive as climates change. Environmental monitoring (e.g., *via* sentinel sites) could help identifying these (**section 5.4.3**). Future sources of invasive alien species are also likely to differ from current sources under climate change, as geospatial matched climate changes across the globe (**Chapter 3**, **section 3.3.4**). New source regions and species threats will require prioritization with associated adjustments to pathway management actions.

Adaptive management will be needed to adapt monitoring, decision-making and management under climate (**section 5.4**) and habitat change. There is the possibility that climate change may alter the efficacy of existing successful species-based management programmes (e.g., biological control; Y. Sun *et al.*, 2020). This may require the development of new management practices to ensure that new control programmes, or gains made during current control programmes, are not impeded. Site-based management priorities may have to be reconsidered based on changing climate (**Chapter 3**, **section 3.3.4**) and reduced resilience of habitats to invasive alien species. The most vulnerable sites being offshore and mainland islands, mountain tops and coastal environments will be critical for supporting threatened and endangered species. The IUCN recognizes the likely increased use of species translocations to save endangered species from declining climate niches and has produced guidelines to support this, but there are risks and consequences (Webber & Scott, 2012; Lozier *et al.*, 2015). Integrating biological invasions management strategies into assisted translocation actions under climate change could help avoid unintended consequences (Webber & Scott, 2012; IUCN/SSC, 2013).

Relevant stakeholder community actions as recommended by the CBD (2022b) include:

- engaging all sectors including agriculture and public health agencies and industries in invasive alien species planning where climate change risks are cross-sectoral;
- raising public awareness, including with local and Indigenous Peoples and local communities, of changing invasive alien species threats arising from climate change and include the participation of the public and all relevant sectors in response planning;
- minimizing the potential of biological invasions or develop spatial response planning for areas in which communities are threatened with a high risk of extreme weather events (e.g., relocate zoos, botanical gardens, aquaculture facilities using alien species, from extreme-event-prone areas).

Currently, invasive alien species are considered under the 2030 Agenda for Sustainable Development only in the context of the terrestrial environment (Sustainable Development Goal (SDG) Indicator 15.8.1), but under climate change it will need to be considered equally in marine environments. Climate change and habitat transformation interactions with invasive alien species at various stages of the biological invasion process are illustrated in **Figure 5.27**.

5.6.1.4 Management challenges in urban areas and coastal developments

Urban and peri-urban environments are the fastest growing ecosystems on earth and provide easy opportunities for invasive alien species introductions (Gaertner *et al.*, 2017); (**Chapter 2**, **section 2.5.5.1**; **Chapter 3**, **section 3.2.2.4**). There are four recognized zones of urbanization in most cities, and urban areas often have different microclimates from the surrounding countryside (Erz, 1966). They are heterogeneous and highly complex human-made ecosystems influenced by strong social and political drivers (Cadenasso & Pickett, 2008). Natural spaces within urban areas are critically important for some constituents of good quality of life such as physical and psychological health. These nature refuges, parks and gardens are still largely human-designed or disturbed so acceptance of some alien species (**Glossary**) is to be expected, which needs to be recognized by management frameworks (Gaertner *et al.*, 2016). Urban environments are often close to country ports of entry, so they experience high propagule pressure from

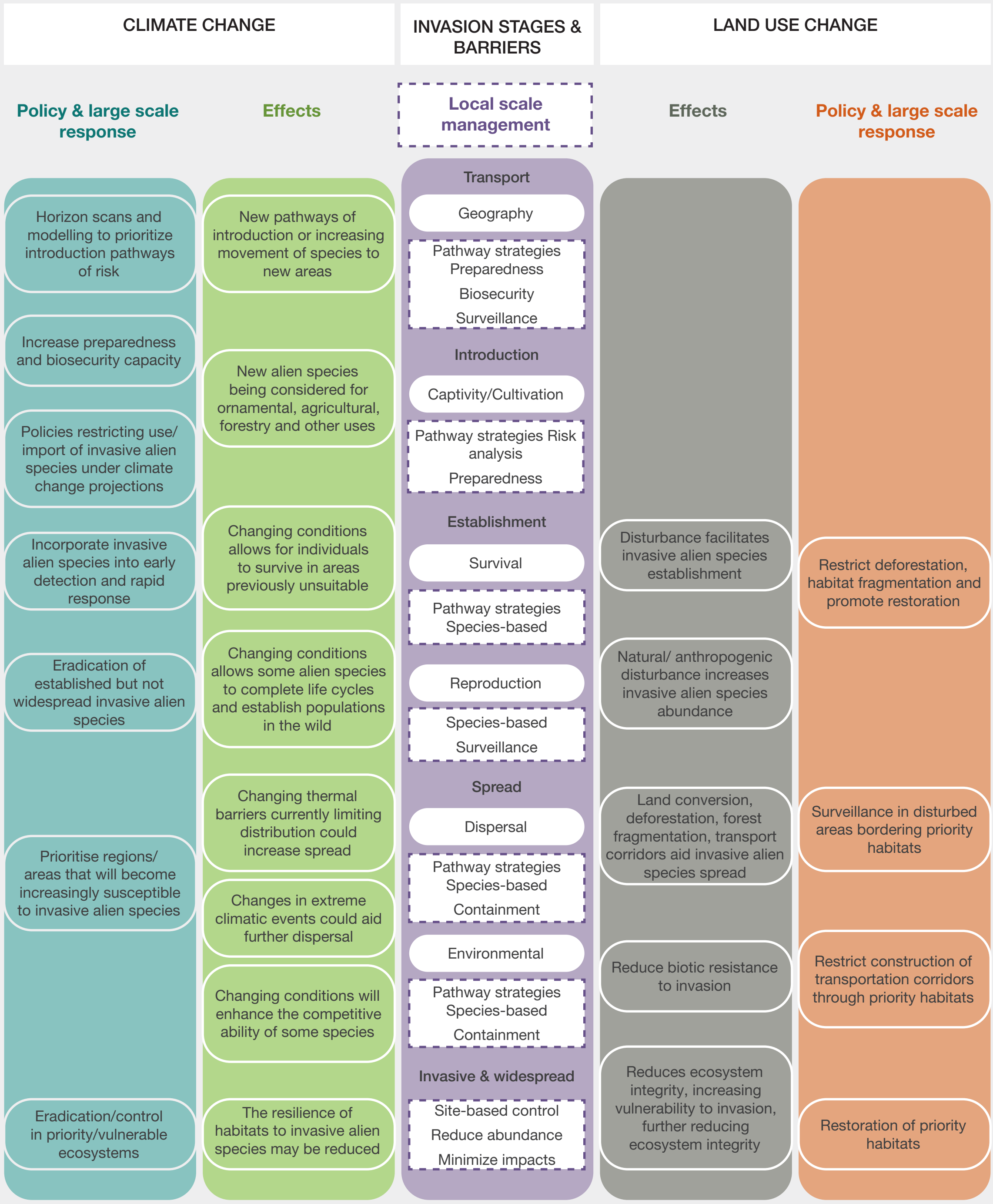


Figure 5 27 **Climate change and habitat transformation interactions at various stages of the biological invasion process, and potential management responses.**

Adapted from Walther *et al.*, (2009), https://doi.org/10.1016/j.tree.2009.06.008, under copyright 2009 Elsevier Ltd. See also **Chapter 3**, **section 3.4.2**.

alien species and provide direct pathways between urban centres globally (Elmqvist *et al.*, 2013; Weiss *et al.*, 2018). Urban areas form a nexus (**Glossary**) of railway and road networks, which are pathways for a wide range of invasive alien species (Ascensão & Capinha, 2017). Being generally highly disturbed, urban areas are therefore susceptible to alien species establishment and the high levels of human activity facilitate spread. One study showed that on average 28 per cent of the flora of the urban areas globally comprise of non-native species (Aronson *et al.*, 2014).

Most intentionally introduced species are alien garden plants, which along with released pets, predominate in the urban and peri-urban settings to which they may be pre-adapted. A global review found that most alien species are intentionally introduced in cities and were either released or had escaped from confinement (Padayachee *et al.*, 2017). Many widespread invasive alien species are well adapted to human landscapes, but some are so well adapted that they may not spread any further (e.g., domestic pigeons; Erz, 1966).

The peri-urban fringe of cities is increasingly interweaved into surrounding agricultural and natural ecosystems, allowing much more intimate interactions between people and wildlife (Seto *et al.*, 2013). This, for example, has recently been recognized as increasing the risk of emerging zoonotic diseases (Di Marco *et al.*, 2020) and the need to invest in pandemic prevention (Dobson *et al.*, 2020). Even though many introduced alien species may not have any perceived impacts in urban areas (McLean *et al.*, 2017), they can spread beyond city limits and invade natural and semi-natural habitats as a growing number of protected landscapes and marine reserves fall within a matrix of broader land-use types or zonation (Seto & Shepherd, 2009). Natural areas surrounded by urban areas also suffer from higher-than-normal propagule pressure from the urban areas. For instance, numerous new city reserves suffer from arthropods, dogs, cats, livestock (e.g., goats) and alien plants that live in close association with humans, the incursions of which reduces the resilience of these ecosystems (Lacerda *et al.*, 2009; Lessa *et al.*, 2016; Paschoal *et al.*, 2016; Spear *et al.*, 2013). Management of biological invasions in urban contexts is especially challenging because on the one hand urban environments to a degree actively encourage alien species for physical, cultural and political reasons, but natural areas connected to them are most threatened by invasive alien species from higher levels of human activity spilling over from nearby urban areas (Gaertner *et al.*, 2017).

Continuous monitoring of urban biota improves early detection of potential invasive alien species (Paap *et al.*, 2017). This is best done by assessing alien species impacts to distinguish non-invasive and invasive alien species as this is critically important for management so resources are used cost-effectively. Urban forests have been used as sentinel sites for the detection of *Agrilus planipennis* (emerald ash borer) helping managers to manage outbreaks early (Poland & McCullough, 2006). *Euwallacea fornicatus* (polyphagous shot-hole borer) and an associated pathogen (*Neocosmospora euwallaceae*) were first detected in urban environments in United States (California), Israel and South Africa, threatening to spread into nearby plantations and native forests (Paap *et al.*, 2020). An advantage of urban areas is the high availability of human support for invasive alien species management through citizen science-based surveillance, detection and rapid response activities (**section 5.4.3.2**). Citizen science initiatives have supported the early detection of *Halyomorpha halys* (brown marmorated stink bug) in Europe (Maistrello *et al.*, 2016) and New Zealand (Payne *et al.*, 2021). The widespread access to smartphones carrying biodiversity or pest recording platforms (e.g., iNaturalist and SIS-Geo) supports these activities giving the entire population the potential as detectors of invasive alien species (New Zealand MPI Biosecurity 2025; Bejakovich *et al.*, 2018; **sections 5.4**, **5.5**).

Coastal and associated off-shore infrastructures offer novel habitats for both biodiversity and the establishment and spread of marine invasive alien species (**Chapter 3, section 3.3.1.4**; Airoldi & Bulleri, 2011; Bulleri & Airoldi, 2005; Giachetti *et al.*, 2020). The total marine surface created by marine construction (as gas and oil platforms, aquaculture and wind farms, recreational and commercial ports, wave and tidal farms, breakwaters, shipwrecks, artificial reefs) was 32,000 km^2 in 2018 and projected to increase 23 per cent by 2028, which is probably an underestimation (Bugnot *et al.*, 2021). Ocean infrastructure is developing faster than marine spatial management and planning, which is struggling to include management of biological invasions, even though eco-engineering may provide solutions (Dafforn, 2017).

5.6.2 Gaps and impediments to implementing management

5.6.2.1 Societal and social impediments to effective management

The likely number of attempts to manage invasive alien species compared to the relatively few that are reported as successful could imply that most attempts did not provide long-term success with objectives achieved. But this need not always be true and invasive alien species management success rates generally increase as decision-making and stakeholder engagement improves and best practice is understood and adopted (**sections 5.2.1**, **5.2.2.1**). For example, there have been high rates of success in invasive alien species eradication on islands (**section 5.5.3**) and in classical insect and weed biological control programmes (**section 5.5.5**). Failures of management can result from numerous procedural, societal and capacity-related constraints (**Table 5.10**; Day & Witt, 2019), as for example the absence of long-term funding necessary to achieve the goals and avoid reinvasions (Dana *et al.*, 2019). These constraints, and the fact that in many cases there is a lack of understanding or the drivers of change that favour the introduction of invasive alien species are not identified (**Chapter 3**), make it difficult to implement pathway, species-based and site- or ecosystem- based management (**section 5.3**), thereby obstructing effective prevention,

Table 5 10 **Identified constraints of effective management that impede successful management at each management approach (pathway, species-based and site/ecosystem based) explained in this section.**

Dark grey cells indicate when there are constraints to effective management approaches.

Constraints		Most affected management approaches		
		Pathway	Species-based	Site/ ecosystem based
Procedural	Jurisdictional boundaries			
	Policy inadequacies			
	Stakeholder engagement			
Capacity-related	Lack of expertise			
	Inadequate communication			
	Resourcing			
Societal	Resistance to management approaches and technologies			
	Lack of awareness			

eradication, containment and control of invasive alien species. Since invasive alien species are a human-caused issue, the constraints described in this section are generally related to the context of the values and perceptions on invasive alien species.

Jurisdictional boundaries: As invasive alien species are impervious to human-created political and legal borders, addressing cross-border invasion (be it property, local, national or international borders) in a cooperative manner is difficult from both a legal responsibility and financial liability perspective. Examples are: a) eradication of the invasive alien species *Hemitragus jemlahicus* (Himalayan tahr) in New Zealand was considered no longer possible because of private legal property rights preventing enforcement of an official eradication programme (Forsyth & Tustin, 2001) and b) in Ireland, a number of protected sites under the European Habitats Directive (Special Areas of Conservation) span the border between the Northern Ireland (non-European Union) and the Republic of Ireland (European Union) (Stokes *et al.*, 2006). There is no formal mechanism for coordinated cross-border control for managing biological invasions, effective management of invasive alien species is challenging even within individual protected areas.

Policy inadequacies: Contradictory or inadequate legislations, policies and regulations are a very common impediment to management of biological invasions. From a prevention perspective, regulating which species can and cannot be introduced live into a jurisdiction is the first line of control (Garcia-de-Lomas & Vilà, 2015). Regulated species lists are lists of species that are either allowed or not into a country. Unregulated species can generally be imported live across jurisdictional boundaries without control. Countries generally ban certain regulated species and allow all others to be imported (most countries), or regulate which species can be imported and ban all species that are unregulated (Australia and New Zealand). In the latter context, for a species to be regulated for import it will need to have been approved under an independent import risk assessment process (**section 5.2.2.1**). Regulating banned species only is a reactive approach while regulating species for import is a proactive biosecurity approach (Burgiel *et al.*, 2006). Allowing any unregulated species (implies no import risk assessment has been undertaken) to be imported creates a major invasive alien species biosecurity risk (Hulme *et al.*, 2018; Simberloff, 2006). At the post-border stage, regulatory power of policies can be an issue of concern. For example, in the United States, most authorized invasive plant lists do not carry any regulatory weight against the use of listed species (Niemiera & Holle, 2009), thereby allowing release and escape of many invasive alien species except for those banned by federal or state governments.

Also, several developing countries lack comprehensive relevant legislation for biological invasions, and even fewer have recognized lists of invasive alien species and an associated regulatory system (Banerjee *et al.*, 2021).

For example, Indonesia has an existing invasive alien species National Strategy and Action Plan built on risk analyses and management priorities, however, it is not supported by effective cross-sector policy regulating importation and movement of invasive alien species, and public understanding of the issue is also inadequate (Setyawati *et al.*, 2021). Such socio-political realities often lead to governmental lethargy even for invasive alien species with actual or potential impacts to the economy (Nuñez & Pauchard, 2010; Zenni *et al.*, 2017; K. Gupta & Sankaran, 2021) and in some cases such species are seen only as potential economic opportunities (Hänfling *et al.*, 2011). This clearly has direct and indirect implications for implementation of effective management. When governments propose tighter regulations on invasive alien species, industries that sell invasive alien species as products lobby against this on the basis that business generates tax benefits for governments (Hulme *et al.*, 2018; Mack *et al.*, 2000), for example, horticulture (Niemiera & Holle, 2009). Most countries are, however, now parties to the CBD which supports national invasive alien species legislation under the Kunming-Montreal Global Biodiversity Framework (CBD, 2020a).

Stakeholder engagement: In most countries important stakeholders related to management of biological invasions are disconnected from the problem. But, identifying stakeholder responsibilities and engaging them in the management of biological invasions are key to successful outcomes (Kamigawara *et al.*, 2020; **Chapter 1**, **section 1.5.1**; **Chapter 6**, **section 6.4**). A comprehensive study investigating the development and sales of alien pasture plants in eight countries located across six continents showed that the vast majority of agribusinesses in these countries, as well as government agencies and other private companies, do not manage the risk of their products; agribusiness could integrate risk analysis with development of new products and avoid trading species which have a high risk of invading natural areas (Driscoll *et al.*, 2014). A similar situation was also reported in ornamental horticulture industry (Niemiera & Holle, 2009). This lack of recognition of social responsibility is at least partly due to a lack of incentives (as they do not see this as their problem) and legal responsibilities (Driscoll *et al.*, 2014; Simberloff *et al.*, 2005). Coordinated participation of private landowners is also crucial, but is often lacking in the actual management (Drescher *et al.*, 2019; Meier *et al.*, 2017) frequently due to lack of legal responsibility or incentives (Epanchin-Niell & Wilen, 2015), hindering effective management of biological invasions across entire landscapes (Glen *et al.*, 2017). There are however many examples where effective regulatory, social responsibility and incentive-based systems support effective industry (Harrington *et al.*, 2003; Burt *et al.*, 2007; Conser *et al.*, 2015) and landowner (G. R. Marshall *et al.*, 2016; Niemiec *et al.*, 2017) engagement in prevention and management of biological invasions.

Lack of expertise: Declining numbers of specialist taxonomists and a shortage of invasive alien species management specialists creates capability impediments to management of biological invasions (e.g., Pyšek *et al.*, 2013) and regulations at policy level (Hieda *et al.*, 2020), leading to errors in decisions in many cases (Bortolus, 2008). This is also a weakness for understanding the status and trends (**Chapter 2**) and impacts (**Chapter 4**) of invasive alien species. *Sporobolus densiflorus* (denseflower cordgrass) introduced in California from south-eastern America was confused with *Sporobolus foliosus* (California cordgrass) and ecosystem restoration activities along the west coast of North America led to its spread (e.g., present in 94 per cent of the Humboldt Bay) until 1985 (Bortolus, 2008; Kittelson & Boyd, 1997) preventing effective eradication (Pickart, 2012). In Australia, *Asterias amurensis* (northern Pacific seastar) was also confused with the native species *(Uniophora granifera)* and by the time this was realized, eradication and control were no longer possible (M. L. Campbell *et al.*, 2007). In Kerala, India, the invasive tree *Senna spectabilis* (whitebark senna) was misidentified as the native *Cassia fistula* (Indian laburnum) and widely planted, and is now widespread in the Wayanad wildlife sanctuary and in the Nilgiris causing impacts to natural and planted forests and coffee plantations (Vishnu Chandran & Gopakumar, 2018). In developing countries, lack of capability in the use of effective prevention and management approaches for biological invasions, tools and technologies are severe impediments to implement better management approaches and could be a focus of international support and aid programmes

Resourcing: Seeking adequate resources to undertake effective and sustainable management of biological invasions is a global problem, while many resources aree being used ineffectively (Courchamp *et al.*, 2017). Making the case for investment requires a) evidence-based economic, social and environmental impact analyses (**Chapter 4, section 4.1.1**) and b) demonstration that any particular invasive alien species management approach provides cost-effective and sustainable mitigation of these impacts, in competition with other government investment priorities. The lack of success of many invasive alien species management programmes does not help this case (Latombe *et al.*, 2019). Some management approaches such as biological control have a long history of cost-effectiveness (**section 5.5.5.3**) and create a strong case for investment. This will need to be demonstrated for the new technologies under development. Developing national capability and capacity for management of biological invasions is also linked to sustained funding (Nuñez & Pauchard, 2010).

Resistance to management approaches and technologies: Public opposition to uncertainty, often resulting from a poor understanding of management approaches for biological invasions and technologies, is a significant impediment to effective management (**Chapter 3**,

sections 3.3.2.4, **3.3.5.2**). Classical biological control, despite a long history of development and benefits and support by both the IPPC and the CBD still attracts negative views (Downey & Paterson, 2016). This is based on historic evidence of non-target impacts (Carvalheiro *et al.*, 2008; Pearson & Callaway, 2005; Willis & Memmott, 2005), some from a time before regulations under internationally agreed risk analysis (Howarth, 1991; **sections 5.5.5.3**, **5.5.5**). Acceptability is impeded by a general lack of government investment to monitor non-target impacts (Barratt *et al.*, 2021; Simberloff *et al.*, 2005). Pesticide-based chemical control is also becoming less acceptable related to short term efficacy and non-target effects (**section 5.4.3.2**; Simberloff *et al.*, 2005). Similarly, opinions on the use of lethal control of invasive alien vertebrates vary widely based on country and stakeholder groups, with the ethical consideration thereof now of high importance (**Chapter 1**, **section 1.5.3**; **section 5.4.3.2**). This is not helped by the many baiting and culling programmes that are poorly planned/implemented or unsustainably resourced. Some Indigenous Peoples and local communities have a moral dilemma about using chemical and biological control methods to manage invasive alien species, because the methods can be incompatible with their spiritual connections with the land (IPBES, 2022a). Animal welfare constraints can also arise from opposing public perspectives on invasive alien species (Estévez *et al.*, 2015), which is detailed in **section 5.6**.

Inadequate communication and lack of awareness: Lack of understanding of invasive alien species impacts (Kleitou *et al.*, 2019) and linguistic problems in communication can be a constraint in management planning across culturally different stakeholder groups, but can be addressed through co-developed communication planning. Stakeholder groups are also likely to be more engaged and committed to implement management strategies in ecosystems they use, which may lead to a bias in management towards terrestrial ecosystems (Mungi *et al.*, 2019). Sosa *et al.* (2021) suggested that support to manage biological invasions can be enhanced by promoting communication between educators and teachers, which will encourage public participation in the process. They also proposed increasing awareness among students by including invasive alien species identification and their potential threats in educational curricula from Kindergarten to University levels (Sosa *et al.*, 2021; **Chapter 6**, **section 6.7.2.4**).

5.6.2.2 Knowledge and implementation gaps in the management of biological invasions

This chapter has identified gaps in the implementation of knowledge, or knowledge gaps that constrain successful long-term control, in pathway, species-based and site/ecosystem-based management of biological invasions. Addressing the gaps identified below can directly support improved management actions, in cases providing the minimum information for decision-making. Alternatively, while certain tools and methods have been developed, how to use them in a particular scenario or at a large enough scale, is not currently known. A summary of these is presented in **Table 5.11**.

Table 5 11 **A summary of gaps in knowledge and implementation impeding management of biological invasions.**

The gaps were developed through an expert elicitation process with authors of **Chapter 5**.

Gap type and category		Gap description	Why is it important?	Cross-reference
Pathway Management	**Knowledge and implementation;** potential instruments, including policy and enabling approaches	Eradication strategies and guidelines for generalist invasive alien invertebrates, diseases and hard to detect freshwater and marine invasive alien species (not restricted to defined hosts).	These groups have been understudied. Even where information is available, developing and implementing guidelines remains difficult and is seldom done.	5.2.2.1, 5.2.2.2, 5.5.3
	Knowledge; gaps on biomes, units of analysis or taxonomic gaps	Risk management, cost-effective species-based surveillance and detection strategies for multiple invasive alien species groups, e.g., fungi and other microbes.	Species-based approaches are limited by taxonomic un- certainty, e.g., microbes. Strategies are needed at a higher taxonomic level than species in such cases.	5.2.2.1, 5.3.1.2, 5.4.3.2
	Implementation; potential instruments, including policy and enabling approaches	Risk analysis for movement of marine invasive alien species.	Risk analysis tools are available but not consistently applied. Pathway management is the highest priority for marine species.	5.2.2.1, **Figure 5.4**
	Implementation; potential instruments, including policy and enabling approaches and management	Managing alien species movements and biosecurity risks along trade supply chains, e.g., *via* shipping containers.	Trade based pathways such as shipping containers and illegal mail order remain poorly managed, particularly for contaminating pests and diseases.	5.3.1.1, 5.4.3.1, **Box 5.2**

Table 5 11

	Gap type and category	Gap description	Why is it important?	Cross-reference
Pathway Management	**Implementation;** potential instruments, including policy and enabling approaches and management	Effective management and compliance of biofouling policy.	International (and national) policy instruments are available but not consistently applied. New biofouling treatments are needed.	5.5.1, Chapter 6, section 6.2.1(5)
	Implementation; management	Management of deliberate movements of species across jurisdictional land-borders. Domestic quarantine is poorly implemented in several developing countries.	Needs better policy to support management. Natural pathways cannot be prevented, but may benefit from improved surveillance.	5.6.2.1, **Table 5.10**
	Knowledge; integrated scenarios and models; technical development	Understanding of direct and indirect non-target impacts of chemical, manual, mechanical and biological control of an invasive alien species on other species and ecosystems.	Non-target impacts can be substantial and are important therefore data need to be collected and included in risk analysis.	5.5.5
Species-based Management	**Knowledge;** gaps on biomes, units of analysis or taxonomic gaps	Incorrect taxonomic species identification (or varieties) impeding management.	Access to strong taxonomic capability for invasive alien species in all key groups is critical.	5.4.3.2, 5.6.2.1, **Table 5.4**, **Table 5.12**
	Knowledge and implementation; integrated scenarios and models; technical development	Prioritizing invasive alien species management and developing the necessary strategies under climate change and habitat or land-use change.	Considering climate change effects on invasive alien species and their management is rare but will be critical in the future.	5.6.1.3, **Figure 5.27**; Chapter 6, section 6.7.2.2
	Knowledge and implementation; integrated scenarios and models; technical development	Prioritizing management of biological invasions over other actions (e.g., threatened and endangered species protection and management).	Protecting threatened species and communities may be improved by understanding cost-effectiveness of different actions including management of biological invasions to prioritize investments.	5.2.2.2, 5.3.1.4
	Implementation; management	Containment of slow spreading pervasive invasive alien invertebrates and plants.	Slow spreading invasive alien species are often a lower priority for management but they may be harder to control later and have greater long-term impacts.	5.5.4
	Implementation; technical development	Humane management approaches for invasive alien species subject to animal ethics.	Humane management approaches for invasive alien species often increases social acceptability.	5.4.3.2, 5.5.5.2
	Implementation; integrated scenarios and models; technical development	Management of invasive alien invertebrates and plants under increasingly restrictive chemical control options.	With the preference to alternative management options, it is important to proactively consider and develop better integrated management approaches including biological options.	5.4.3.2, 5.4.3.3
	Implementation; management	Management of marine invasive alien species for population suppression.	All current marine invasive alien species management programmes have been unsuccessful in the long-term as a means of control.	5.6.1.1, **Box 5.3**
	Implementation; management	Management approaches for widespread established invasive alien species using available and novel tools and methods.	Once prevention has been optimized there is a need to consider and develop better technologies for control of widespread species.	5.5.5
	Implementation; integrated scenarios and models	Management decision-making approaches for invasive alien species with benefits in some contexts (i.e., conflict species).	Policy and collective decision-making approaches need to better address conflict species to prevent management being stalled.	5.6.1.2, **Table 5.9**; Chapter 6, section 6.4.1
	Knowledge; integrated scenarios and models	Prioritizing site-based management under multiple management contexts (i.e., nature, nature's contributions to people and good quality of life).	Site-based, ecosystem-based and restoration generally focuses on biodiversity protection but needs to include impacts on Indigenous Peoples and local communities.	5.3.1

Table 5 11

	Gap type and category	Gap description	Why is it important?	Cross-reference
Site/ecosystem-based Management	**Knowledge;** integrated scenarios and models; technical development	Cost-effective scenarios and modelling for invasive alien species management and evaluation use.	Scenarios and modelling are generally underutilized for invasive alien species management planning.	5.2.2.4, 5.6.3.2, **Table 5.14**, **Table 5.15**
	Knowledge; potential instruments, including policy and enabling approaches and management	Managing urban and peri-urban areas, including urban-marine linked areas, in the context of impacts on surrounding ecosystems and ecosystem services on which local communities depend.	As urban and peri-urban areas put increasing pressure on native communities through local biodiversity loss, managing this driver of invasive alien species impacts needs to be prioritized and addressed.	5.6.1.4
	Implementation; management	Effective inclusion of Indigenous and local knowledge in management design and decision-making.	Indigenous and local knowledge is critical for long-term, integrated, management of biological invasions.	5.1.3, 5.5.2, 5.7, **Box 5.15**
	Implementation; management	Adaptive integrated invasive alien species management with ecosystem restoration to improve ecosystem resilience and broader ecosystem-based management.	Improving adaptive management from governance to implementation is a priority, as it is a proven approach to managing dynamic ecosystems.	5.4.3.3, 5.5.6, 5.6.2.5; Chapter 6., Box 6.14 and Box 6.16
Other implementation gaps	**Essential supporting processes as impediments to invasive alien species management;** potential instruments, including policy and enabling approaches and management	Procedural (policy, cross-jurisdictional, stakeholder engagement). Capacity-related (capability, lack of knowledge on modern tools and techniques, resourcing and communication). Societal (lack of awareness, resistance) challenges will need to be addressed.	Biosecurity and invasive alien species have a human cause, are a function of human values and endeavour, and therefore need greater cooperation and social and societal analysis and solutions.	5.6.3.3, **Table 5.10**; Chapter 6, sections 6.4 and 6.7
	Uncertainty; integrated scenarios and models; technical development	Decision-making in the context of uncertainty.	The precautionary approach argues that actions should not be hampered by incomplete knowledge where doing nothing is not an option.	5.2.2.3, 5.6.2.5

5.6.2.3 Challenges to management in relation to knowledge gaps in invasion biology

While technological advances (**section 5.4**) are assisting management of biological invasions, it is critically important that ecological understanding at the species and community levels (Zavaleta *et al.*, 2001) and Indigenous and local knowledge underwrite their application to avoid perverse outcomes (e.g., Caut *et al.*, 2009; **section 5.5.4**). Since biological invasions are non-linear and dynamic (**Chapter 6**, **section 6.6.1.1**) the resulting complexity needs to be recognized when preventing and managing biological invasions around the world. There are multiple examples of invasive alien plant replacements following mis-informed management that can make the system worse (Pearson *et al.*, 2016). Invasive alien vertebrate management programmes can also lead to unexpected consequences. Invasive pig control in Hawaiian rainforest removed a disturbance agent supporting native species recovery in some areas (Loope *et al.*, 2013), but led to a five-fold increase in the invasive alien plant *Psidium cattleianum* (strawberry guava) in others (Kellner *et al.*, 2011). Similarly invasive cat suppression can allow invasive alien rodent densities to increase (Karl & Best, 1982; Zavaleta *et al.*, 2001). In Kakadu National Park (Australia), expansion of *Urochloa mutica* (para grass) expansion followed invasive *Bubalus bubalis* (Asian water buffalo) removal (Morris, 1996; **Chapter 3**, **section 3.3.5.2**). Monitoring biodiversity and freshwater ecosystems using macroinvertebrate-based indices is a widely-used method globally, however knowledge is lacking on how invasive alien species may affect the metric scores and therefore classification of a river's status (Guareschi & Wood, 2019).

5.6.2.4 Insufficient technological expertise in implementing management techniques

Based on the available information, the numbers of invasive alien species in developed countries are significantly higher compared to developing countries (IPBES, 2019). However, this could be an incorrect assertion since several developing countries are underexplored for invasive alien species and/

or data are unavailable, especially for some ecosystems (e.g., for marine ecosystems), resulting in significant data gaps. As a result, invasive alien species are poorly managed or unmanaged in these regions (McGeoch *et al.*, 2010). This may also be due to the gaps in capacity and capability (i.e., expertise and experience) between developed and developing countries in management of biological invasions. Technologically advanced countries may provide strategies and solutions through aid programmes to developing countries, but aid programmes rarely have the long-term support to ensure systemic adoption (Boy & Witt, 2013) and generally fail when development of local expertise is not supported, not adequately co-designed, institutionalized and resourced. This creates a huge impediment for the adoption of the many effective tools and technologies currently available (**section 5.4**) as many regions are unable to utilize them. Local communities' distrust of unfamiliar techniques is also one of the impediments for adoption (**sections 5.4**, **5.5**). For monitoring, lack of resources or skills precludes adopting many advances in technology such as environmental DNA, remote sensing, or the use of unmanned aerial vehicles for invasive alien species detection (**section 5.4.3.1**). Technological solutions need to be set in the local context encouraging local communities to adopt them in a manner applicable to their conditions, experience and resourcing. Adoption of autonomous technological solutions in developing countries has been effective in other sectors (e.g., vaccine delivery in Ghana). In many regions, the use of pesticides is disallowed due to lack of regulations and for fear of non-target effects. For weeding in agricultural areas, which frequently includes invasive alien plants, this leaves manual removal as the only option. In Africa and the Asia-Pacific region manual removal is the most time consuming and costly activity for local farming communities (Day, Kawi, Tunabuna, *et al.*, 2011; FAO, 2006; Muraleedharan & Anitha, 2000; Sims *et al.*, 2012). In such cases, the use of classical biological control may provide long-term solutions, however aversion to such techniques needs to be overcome (Boy & Witt, 2013). Sharing of technological expertise to manage biological invasions can be achieved with international cooperation and by building long-term relationships (Hulme, 2020b; **section 5.6**; **Chapter 6**, **sections 6.3.1**, **6.6.2.2**).

5.6.2.5 Applying adaptive management under uncertainty

Adaptive management is a key approach in management of biological invasions (Foxcroft & McGeoch, 2011; Zalba & Ziller, 2007), assisting decision-making in management where there are data and knowledge gaps (**Chapter 6**, **section 6.6**, **Figure 6.20**). This is usually the case when resources are limited, and the management system is socially and ecologically complex (**sections 5.4**, **5.5**). Uncertainty often leads to the tendency of inaction and delaying management actions for want of complete information (Salafsky *et al.*, 2001), however, this needs to be compared to the consequences of inaction. In management of biological invasions, some decisions and actions need to be taken rapidly, for example, to manage a pathway during an unexpected incursion, or to initiate species-based eradication while it is still feasible (S. Liu *et al.*, 2011). Therefore, management decisions need to be made and actions implemented based on the best available knowledge (Stohlgren & Jarnevich, 2009). Accurate information is a precondition for undertaking effective and timely management measures, including species identification, gained from field and literature surveys (e.g., Island Conservation, 2018). Errors often lie across the scales and types of invasive alien species data with potentially serious consequences for prevention and management (McGeoch *et al.*, 2012). Examples are given in **Table 5.12**. Field validation of knowledge is important and can be obtained during the management implementation using an adaptive management approach.

Table 5 12 **Impacts of errors in data on invasive alien species presence, distribution, socio-economic and political perceptions and potential responses to improve the efficacy of management interventions on biological invasions.**

Adapted from McGeoch *et al.* (2012). See **sections 5.4** and **5.5** for details on management methods.

Type of error	Explanation	Effect on management or policy development	Management responses (instruments tools and approaches)
Data collection	There can be a lack of survey information on the presence, extent, and population dynamics outside the native range of a species. Resolution of data and scaling in the introduced range of the invasive alien species: he low-resolution of alien species distribution maps or geographic regions can lead to overestimation of species distribution.	a) Data on the establishment and spread is required to designate alien species as invasive. Insufficient survey information results in failure to recognize invasive alien species. b) Invasive alien species assemblages are dynamic, and the lack of regular surveys can lead to inaccurate species lists and data on distribution and population sizes.	Increased attention to detail and taking care to record data correctly, and increasing efforts to search for information to ensure correct species identification (including synonyms, name changes, incorrect names). Increased frequency of data surveys for a better recognition and definition of invasive alien species distribution

Table 5 12

Type of error	Explanation	Effect on management or policy development	Management responses (instruments tools and approaches)
Data collection		c) Populations may be incorrectly delimited (prevalence known) leading to incorrect decision-making and management errors. d) Prematurely declaring eradication campaigns successful when not enough monitoring has been done to ensure confidence in eradication as cryptic populations remain un-detected.	(**Chapter 6**, **sections 6.6.2.4** to **6.6.2.7**).
Data and knowledge not documented or not readily or widely accessible	Data are not available in books and peer-reviewed literature, electronic, or online databases. Information may exist (and specialists may recognize invasive alien species), but is not yet documented, or is outdated. Grey literature is not easily accessible and may be in different languages. Some of the new taxa data are published in obscure journals. A wide range of data sources exist, but are not always sufficiently well collated, published or easily accessible.	This may result, for example, in a time delay between discovery and publication. This may influence the likelihood of eradication opportunities. Eradicated or extirpated species may also remain on species lists. Inadequate native range information (e.g., cryptogenic species – see **Glossary**), may result in subjective or incorrect listing of species as being alien or not. Identifying an alien species incorrectly, a lack of information on how to implement management, and a lack of specific/appropriate management tools.	Enhance connectedness of global repositories (**section 5.4**), especially for data and grey literature (**section 5.4**). Use of taxonomic expertise (Pyšek *et al.*, 2013) and identification tools to assist in correct species identification (**section 5.4**; **Chapter 6**, **section 6.6.2.2**).
Incomplete information/ literature searches and species misidentification	Erroneous information in lists and databases may be perpetuated.	Misidentification of species, without recognizing synonyms, changing names and other errors in data entry. Lack of comprehensive information searches can result in incomplete lists. Alien species can be misidentified as a result of lack of taxonomic data, such as undescribed species or taxa where the systematics have not been fully resolved.	Conscientious and thorough reviews and assessments before decision-making (**section 5.2**; **Chapter 6**, **section 6.6.1**).
Socio-economic and perception data	Differing perspectives leading to different perceptions in the community concerning management. E.g., hunters have a vested interest not to reduce density of an invasive alien species of their interest or completely eliminate the target species.	Difficulty to gain consistent perspectives on invasive alien species management directions and planning.	Collaborative and adaptive co-management (**section 5.4**; **Chapter 1**, **section 1.5.2**; **Chapter 6**, **section 6.7.2.4**).
Political perspectives	Political will may vary with different political perspectives and situations.	Management of biological invasions is not a priority item for some countries and may receive only limited/ intermittent funding. Jurisdictional boundaries complicate management responses (**section 5.5**).	Globally, implementing treaties and conventions (**section 5.5**; **Chapter 6**, **section 6.1.3**). Locally, initiatives such as Trans-frontier protected areas or biospheres reserves provide vehicles for collaboration (**section 5.3**; **Chapter 6**, **section 6.3**).

5.6.3 Supporting approaches to improve the uptake of effective management of biological invasions

5.6.3.1 Role of national and international networks and regional partnerships in management

The capacity of governments and resource managers to prevent and manage biological invasions depends on open and quick access to the best available scientific information, data and evidence of impacts (including socio-economic impacts) and access to suitable management tools and approaches (**Chapter 6**, **section 6.6.1**). National and international networks and partnerships are key to achieve these goals (**Supplementary material 5.13**; Fonseca *et al.*, 2013; Simpson, 2004; Simpson *et al.*, 2009; Soubeyran *et al.*, 2015; Wallace *et al.*, 2020) through trust and a feeling of shared responsibility (S. Graham *et al.*, 2019; Nourani *et al.*, 2018). **Table 5.13** presents an overview of common challenges in national and international network development as opposed to local collaborative governance networks focused on active management implementation (**Chapter 6**, **section 6.4.4**). Networks and partnerships also encourage collective efforts which may lead to socially acceptable and feasible strategies for management (S. Graham *et al.*, 2019; Nourani *et al.*, 2018). International conventions and organizations such as IPPC for plant-based trade, the WOAH for animal health, the IMO for shipping pathways (ballast water and biofouling), the CBD for e-Commerce and Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) for movement of protected species help global coordination and cooperation in the management of pathways which is crucial to significantly reduce unintentional introductions *via* international pathways. **Chapter 6**, **sections 6.3.2**, **6.4.4** and **Box 6.9** cover the role of networks and partnerships in managing biological invasions in more detail.

5.6.3.2 Scenarios and models to support the design and implementation of management

Many useful scenarios and modelling approaches have been developed and used to support decision-making and implementation in the context of biological invasions (e.g., Buchadas *et al.*, 2017; Dinis *et al.*, 2020; Gallien *et al.*, 2010; Hall *et al.*, 2021; **sections 5.2** and **5.4**). Indeed, ecological population modelling as a discipline started through the exploration of the predator-prey relationships linked to classical biological control management systems against invertebrate pests (e.g., Hassell, 1978; Mills & Getz, 1996; Murdoch *et al.*, 2013). Since then, species ecological-based and epidemiological-based models have been used in a multitude of forms including deterministic and stochastic models, matrix-based, agent-based and simulation models. These have all actively been used to assist the understanding and management effectiveness of many terrestrial and a few aquatic invasive alien species including plants (e.g., Buckley *et al.*, 2003a, 2003b, 2007), vertebrates (e.g., Calvete, 2006; Garrott *et al.*, 1992; K. Graham *et al.*, 2021; C. M. King & Powell, 2011; McCarthy *et al.*, 2013), plant disease-causing organisms (e.g., Filipe *et al.*, 2012; Harwood *et al.*, 2011) and invertebrates (e.g., Herms & McCullough, 2014). Process-based models, where the physiological or environmental characteristics of the invasive alien species are inputs and underly the model outputs have been used for decision-making and implementation of multiple management actions (Strand, 2000; Sutherst *et al.*, 2011) including biological control (Shea *et al.*, 2002). They are used most frequently in scenario and modelling studies as identified in the literature review to this assessment (107 of 183 studies; **Table 5.14**; Lenzner *et al.*, 2021).[27] These models have supported strategies at national and sub-national level, on

27. Data management report available at: https://doi.org/10.5281/zenodo.5706520

Table 5 13 **Overview of challenges in international and national network development and potential solutions to address them.**

Review and synthesis of S. Graham *et al.* (2019); Groom, Desmet, *et al.* (2019); Katsanevakis *et al.* (2013); Lucy *et al.* (2016); Piria *et al.* (2017); Reaser, Simpson, *et al.* (2020); Simpson *et al.* (2006); and Simpson *et al.* (2009).

Challenge	Solutions
Technical	Co-develop tools for early detection, eradication and control
	Provide high quality, up-to date and accessible data
	Assure interoperability and data standardization
Societal	Raise awareness
	Drive political choices
	Ensure access to funds wherever necessary
Coordination	Overcome national boundaries
	Avoid overlaps and fill gaps in knowledge
	Link thematic group and local networks
	Designate governance and responsibility

terrestrial invasive alien plants and invertebrates. Economic and bioeconomic models have also been developed to support management such as for feral pigs (Zivin *et al.*, 2000), diseases (Petucco *et al.*, 2020) and invertebrates (Marten & Moore, 2011; Vannatta *et al.*, 2012). Modelling has also been applied to the understanding of invasive alien plant management as part of integrated management and ecosystem restoration (Caplat *et al.*, 2012; Firn *et al.*, 2010). However, the consistent use and application of these approaches to support management of biological invasions including decision-making and response actions to prevent or reduce negative impacts is lacking, particularly in marine systems where there are significant gaps.

Correlative models, mostly found in the literature review conducted for this assessment (**Chapter 1**, **section 1.6.7.3**, have only been developed and applied to individual management activities (49 of 183 studies considering management; such as control. These correlative models have principally been implemented on invasive alien invertebrates and mammals from Asia and the Pacific, estimating and quantifying potential impacts and changes in species occurrence or abundance. Correlative models are also often used to build species risk maps both under current and future conditions to estimate invasion potential (e.g., under different climate change scenarios; **section 5.2.2.4**; **Chapter 1**, **section 1.6.7.3**). Hybrid and expert-based models, far less found in the literature (22 and 6 of 183 studies considering management respectively), have been applied especially in Africa, Antarctica, Europe, Oceania and Central Asia, to help assess the prevention and preparedness for the management of biological invasions across many different invasive alien taxa (e.g., amphibians, birds, fungi, fishes, reptiles) in freshwater and marine realm considering an international extent and context.[27] Examples are given in **Table 5.15**. The use of scenarios and models for the management of marine and cryptic species remains challenging due most likely to a lack of environmental data and data on species occurrence that can be used to develop scenarios.

As the process of invasion is dynamic, scenarios and models help make projections to assist, independently or in combination, with preparedness and management goals and decision-making (**Figure 5.1**; **section 5.2.2.4**). Both long (until 2050-2100) and mid-term (until 2030-2050) projections have been obtained under varying management scenarios, but they were little used in retrospective policy evaluations (IPBES, 2016; C. M. Jones *et al.*, 2021). Similarly, projections have been made on single or multiple management approaches addressing one or more drivers in the context of invasive alien species scenarios **(Table 5.14)**. Scenarios supporting management goals have been both qualitative and quantitative, mainly exploratory (109 of 183 studies considering management),[27] and they have principally considered scenarios of management, invasive alien species characteristics (i.e., demographic, dispersal, interaction; 29 publications), climate change (35 publications) and land- and sea-use change (12 publications) as drivers, alone or along with other drivers (**Table 5.14**; **section 5.6.1.3**).

Models and scenarios can be important tools to understand opportunities and contexts of desirable outcomes of management in terms of biodiversity, nature's contributions to people and good quality of life (IPBES, 2016). For example, different scenarios of effective management of rats and an introduced *Philornis downsi* (avian vampire fly), on nesting success of the critically endangered *Camarhynchus heliobates* (mangrove finch) were developed as part of a management programme on Isabela Island in the Galápagos archipelago. These scenarios were used to understand potential management interventions on finch population recovery to identify positive biodiversity outcomes (Fessl *et al.*, 2010). Similarly, an agent-based model was developed for understanding hypothetical agricultural subsidy scenarios aimed at controlling invasive guava and assess the resulting population and land cover dynamics effecting community livelihoods on the same island (Miller *et al.*, 2010). Management options have also been explored to provide positive outcomes to nature's contributions to people (e.g., reduction of economic losses and carbon emissions; Alaniz *et al.*, 2020). Scenarios and models have been poorly used, however, to evaluate the outcomes of management programmes for nature's contributions to people (around 12 per cent of 183 studies) and good quality of life (only 10 per cent of the studies; **Table 5.14**). Scenarios and models with emphasis on Indigenous and local knowledge or Indigenous Peoples and local communities (e.g., with participatory target-seeking scenarios) have been rare (4 per cent of studies focused on management), though involvement of Indigenous Peoples and local communities may lead to better invasive alien species management (**section 5.5**). See **section 5.6.2.1** and **Chapter 6**, **section 6.6.1.6**, for gaps and future directions of scenarios and models which may support management of biological invasions. Scenarios and models have also been used to translate the potential impact of management actions into consequences to good quality of life. For example, for *Aphis glycines* (soybean aphid), an invasive pest of *Glycine max* (soyabean), a dynamic bioeconomic model was developed to estimate the optimal chemical control strategy while considering explicitly the economic value of natural pest control (i.e., the control of the aphid by their natural enemies). Thus, this approach would reduce the economic inefficiency of pesticide use, leading to income security (W. Zhang & Swinton, 2009).

Table 5 14 **Overview of model and scenario types as well as drivers.**

Overall, 183 publications were identified to include management of biological invasions. Numbers for the different groups can be higher than 183 if multiple model or scenario types or multiple drivers were considered in the same study. Evidence from the systematic literature review on scenarios and models undertaken for the IPBES invasive alien species assessment. Data management report available at: https://doi.org/10.5281/zenodo.5706520

	Number papers
Model types	
Expert-based models	6
Correlative models	49
Process-based models	106
Hybrid models	22
Scenario types	
Exploratory scenario	109
Target-seeking scenario	25
Intervention scenario	46
Missing information	3
Drivers	
Climate change	35
Land/sea use change	12
Pollution	1
Demographics	9
Socio-economic development	16
Resource extraction	2
Governance	9
Invasive alien species	29

Table 5 15 **Examples of scenarios and models used for invasive alien species management.**

Biomes	Type of action	IPBES zones	Examples
Terrestrial	Prevention, containment	The Americas	Spatial model screening the economic cost of programmes preventing the spread of satellite populations of an invasive beetle, under different scenarios of success and simulations of spread, versus the estimated delayed cost of control, under different scenarios of actions as removal, and/or no action. Preventing the establishment of new populations is cost-effective (Kovacs *et al.*, 2011).
	Early detection, containment, eradication	Not stated	Economic simulation model evaluating the cost and success of eradication or containment potential actions under different scenarios of detection rates and search efforts (i.e., early detection; Cacho *et al.*, 2010).
	Eradication, containment	Europe and Central Asia	Simulation population model evaluating percentage of reduction in invasive plant species range under different scenarios of removal of individuals, human density and invasive populations characteristics (Wadsworth *et al.*, 2000).
	Eradication	Asia and the Pacific	Stochastic spatio-temporal model evaluating the rate of spread of invertebrates under different eradication scenarios (Kadoya & Washitani, 2010).

Table 5 15

Biomes	Type of action	IPBES zones	Examples
Terrestrial	Containment	Oceania	Process-based model of the impact of climate change on the distribution change of an invasive shrub based on its physiological tolerances for growth and reproduction (Kriticos *et al.*, 2003).
	Control	The Americas	Epidemiological model to understand the capacity for spread of the pathogen *Phytophthora ramorum* (sudden oak death) and the degree to which this is likely to influence management options (Filipe *et al.*, 2012).
		Oceania islands	Matrix-based population model for estimating the population growth rate of stoats to define dulling strategies that will lead to effective population and impact suppression of this introduced predator of ground nesting birds (C. M. King & Powell, 2011).
	Control, biological control	Oceania	Multi-level mixed effects and individual based ecological models allowed management strategy ranking based on potential to suppress population size of the invasive plant *Hypericum perforatum* (St John's wort; Buckley *et al.*, 2003b).
		The Americas	Bio-economic model to develop a general stochastic optimal control framework for the management of an invasive invertebrate using integrated pest management (Marten & Moore, 2011).
	Biological control	The Americas	Deterministic and stochastic ecological population model evaluating the 20-year effective biocontrol of citrus red scale (Murdoch *et al.*, 2006).
	Restoration, management	The Americas	Process-based state-transition model evaluating positive and negative impacts of different restoration scenarios of fire, livestock and grazing and invasion rates of non-native plant species (Forbis *et al.*, 2006).
Freshwater	Prevention	The Americas	Correlative models are used in cost–benefit analyses for prevention efforts, considering various scenarios of lakes at risk of being invaded by crayfish and different actions, from full protection (i.e., all lakes) to few lakes protected. Even with high expenditure on lake protection, net economic benefits were higher (Keller *et al.*, 2008).
	Eradication	Africa	Spatial ecological model evaluating potential management scenarios of pond-breeding frog species considering pond networks, ecotypes (i.e., arboreal, aquatic, terrestrial), access for managers to ponds due land use change (i.e., number of pods targeted) and percentage of individual removal (Vimercati *et al.*, 2017).
	Eradication, Containment	The Americas	Process based model evaluating potential management scenarios that included selective and non-selective removal of fish individuals based on age-group (Chizinski *et al.*, 2010).
	Containment	Asia and the Pacific	Ecological population model evaluating potential management scenarios on abundance of invasive alien species considering river flow conditions for various corridors and containment through commercial fishing or trap removal of individuals (Koehn *et al.*, 2018).
	Control, biological control	Oceania	Correlative hydrological, ecological and epidemiological based spatio-temporal habitat suitability modelling to prioritize future areas for common carp biocontrol in Australia using the virus CyHV-3 (K. Graham *et al.*, 2021).
Marine	Prevention	Europe and Central Asia	Correlative age-base modelling and hydrodynamic models of surface flow are used to evaluate the risks of spreading of fish and invertebrates, associated with intentional or unintentional discharges of ballast water, and considering scenarios of dispersal (i.e., types spreading of groups of organisms) and connectivity (Hansen *et al.*, 2015).
		The Americas	A Bayesian network relative risk modelling is used to detect the areas of a coastal region at greatest risk of invasion. Risk reduction is evaluated under ballast water treatment scenarios considering a decrease in non-native species introductions or their removal after introduction (Herring *et al.*, 2015).
	Eradication; Containment	Not stated	Matrix models are used to explore the efficacy of possible control strategies by removal of crab individuals at critical stage ages and seasons (Z. Zhang *et al.*, 2019).
		The Americas	Correlative models are used to evaluate the success of various fishermen harvest scenarios as control strategies, different levels of interaction complexity among the biotic and abiotic components of the ecosystem and restoration programmes of native species (Ortiz *et al.*, 2015).
		Asia and the Pacific	Process-based spread models are used to forecast areas of potential arrival of invasive crabs through different pathways. These models are complemented with quarantine scenarios preventing transport of crabs by vessels and estimated delayed times of arrival are estimated for areas with greater risk (Koike & Iwasaki, 2011).

5.6.3.3 Biosecurity policy built on prevention and preparedness

The cost-effectiveness of investing in prevention (e.g., pathway management) to thwart introduction and establishment of invasive alien species has been emphasized in this chapter. However, absolute prevention is not always possible and cannot be expected to be successful in all circumstances. It is also difficult to prove that preventative measures are effective, especially when the pathways are diverse and complex. Prevention is therefore best complemented by preparedness, by ensuring that the government, industry and the community are ready when new threats are intercepted, become established and start to spread (e.g., Bacon *et al.*, 2012). Therefore, while preventive measures and biosecurity are essential components of a management strategy, best practice includes investment in preparedness (**section 5.4.2**). Well-developed biosecurity systems in island lead to the interception of large numbers of potentially invasive alien species, which significantly reduces the rates of new introductions (in the case of Australasia exponential introduction rates have been reduced to linear introduction rates; CSIRO, 2020). New introductions are inevitable in any system, but preparedness and associated rapid response strategies help suppress impacts **(Box 5.2)**. In this context, the Australian biosecurity system has recently been forward valued at AU$ 314 billion over the next 50 years, suggesting significant benefits (Dodd *et al.*, 2020).

5.7 CONCLUSIONS

Chapter 5 reviews and assesses the efficacy of various approaches, programmes and tools, to prevent biological invasions and manage invasive alien species and their negative impacts on biodiversity, threatened and endangered species and communities, sociological and economic systems and nature's contributions to people. The chapter focuses on identifying solutions to manage these impacts across ecosystems, species and regions. To do so, the generalized invasion management continuum or invasion curve **(Figure 5.1)** was used to illustrate the progression of invasion from pre-introduction to widespread invasion. This figure also provides insights into potential management actions at different phases of invasion.

The chapter provides evidence that a large body of knowledge and experience already exists for the development and successful implementation of suitable biological invasion management plans at the local, regional and national levels (**sections 5.2**, **5.6.3**). These management plans rely on active engagement and knowledge-sharing of broad stakeholder groups and Indigenous Peoples and local communities in goal-setting, decision-making and intervention through management actions. Since invasive alien species is a human concept, management of these species may cause conflicting values, interests and perceptions for different stakeholders and Indigenous Peoples and local communities (**section 5.1.3**).

Explicit decision-making for the management of biological invasions is transparent, adaptable, repeatable and ensures stakeholder participation, education and endorsement of management choices (**sections 5.2**, **5.6.2**; **Chapter 6**, **section 6.2**). This chapter has presented a range of decision-support tools available for the identification of hazards, the prioritization of pathways, species and sites, the choice of the best management option and the evaluation of progress to achieve the desired outcomes (**section 5.2.2**, **Table 5.6**). Indigenous Peoples and local communities have unique systems for decision-making, from recognition of the need to manage invasive alien species to the choice of management options. The chapter also underlines the need of utilizing both scientific and Indigenous and local knowledge to make the optimal management decisions (**section 5.2.1**).

Limited evidence and/or uncertainty on invasive alien species and their potential or actual impacts are not an obstacle to the implementation of management strategies. Instead, the adoption of a precautionary approach, as appropriate, risk assessments and adaptive management approaches has been shown to provide long-term opportunities (**sections 5.2.2**, **5.3.1**, **5.3.3**, **5.4.3.3**). Management of biological invasions can be achieved by managing pathways, species and site/ecosystem, which can be effectively done individually or in combination, depending on the management goals, the status of invasion, the type of ecosystem and the socio-economic context (**section 5.3.1**). Their combined use fosters more informed decision-making and resource allocation and can be applied at multiple scales (**section 5.3.3**). Of these, pathway management is critical since permeability of pathways may promote introduction and spread of new invasive alien species (**section 5.3.1**).

Many efficient management methods, tools and technologies such as precision application of pesticides, baits, biological control and gene drive technology are increasingly becoming available, and new technology is being rapidly developed (**section 5.4.4**). These methods relate to pathway management (**sections 5.4.2.1**, **5.4.3.1**, **5.5.1**), surveillance and detection, rapid response and eradication (**sections 5.4.2.2**, **5.5.2**, **5.5.3**), containment, local to landscape level management (**sections 5.4.3.2**, **5.5.4**, **5.5.5**) and ecosystem restoration (**sections 5.4.3.3**, **5.5.6**). Scenarios and modelling approaches are important in management of biological invasions to assess management responses, to predict the risk of future incursions and to plan effective eradication-containment–control approaches (**section 5.6.3.2**).

Involvement of all stakeholders is central for planning, decision-making and implementing management programmes for biological invasions, through the promotion of co-implementation, social learning and broad partnerships (**sections 5.2.1**, **5.6.2**; **Chapter 6**, **section 6.2**). Also, stakeholder-led adaptive management involving Indigenous Peoples and local communities promotes wide acceptance and capacity-building and optimization of management success and economic, environmental and social outcomes (**sections 5.2.1**, **5.6.1.2**). Averting this partnership may impact good quality of life of people who are dependent on or have adapted to utilizing invasive alien species as a resource.

Prevention and rapid intervention are the most efficient and cost-effective management approach sustainable in the long-term, which is crucial for marine ecosystems (**section 5.5.1**). However, prevention may not always be successful. National biosecurity systems and invasive alien species legislation can underwrite prevention by ensuring that jurisdictions have suitable regulations and incentives in place and that there is preparedness (surveillance and monitoring) and rapid response capability in the community to address future incursions (**section 5.6.3**; **Chapter 6**, **section 6.3.2**). Surveillance for newly introduced species through citizen science and social media provides broader security by upskilling and engaging the public (**section 5.5.2**). Prevention through pathway management may be successful only if international biosecurity standards are implemented scrupulously (**section 5.5.1**).

Effective eradication approaches have been developed and may be cost-effective only on smaller islands, mountain tops, wetlands and other refuges of high socioecological and biodiversity values (**section 5.5.3**). Eradication elsewhere is unlikely to succeed, unless the incursion is very localized, easy to detect (no hidden propagules), delimited and spreads slowly. There is demonstrable technical know-how to eradicate invasive alien animals on islands, but success depends on a supporting community to implement actions.

Challenges to management of biological invasions include knowledge gaps (**section 5.6.2.1**), inadequate legislation, lapses in implementing the legislation, poor awareness, lack of capacity and capability, know-how and resources (**sections 5.6.2.2**, **5.6.2.4**; **Chapter 6**, **sections 6.6**, **6.5**) and conflict of interests around invasive alien species that are harmful in one context or sector but beneficial in another (**section 5.6.1.2**). These impediments significantly challenge management attempts, especially in the developing economies (**section 5.6.2**).

A comprehensive review of the effectiveness of management of biological invasions leading to measurable improvements in reducing the impacts of invasive alien species on biodiversity and ecosystem services was beyond the scope of this assessment. However, there is strong evidence that management success can be achieved if pathway, species-based and site-based management strategies, identified through evidence-based decision-making, are implemented using appropriate tools and techniques (**sections 5.2**, **5.3**, **5.4**, **5.5**). These approaches may be applied during all stages of the biological invasion process in the proper context and depending on the types of invasive alien species and ecosystems (**sections 5.1**, **5.5**). Adaptive management strategies assist the process supported by stakeholder engagement and Indigenous and local knowledge (**sections 5.4.3.3**, **5.6.2**). The outcomes of these integrated approaches will provide maximum benefits for nature, nature's contributions to people, including the economy, good quality of life (**section 5.4.3.3**).

Chapter 5 has taken a positive and solutions-focussed approach. Although the vast majority of actions taken to manage biological invasions around the world over the last 70 years have not provided long-term success, this chapter did not undertake a complete and objective review of successes *versus* failures. Chapter 5 recognizes that, around the world, failures have led to continued learning and this has resulted in an increasing number of successful programmes and outcomes. Successful results come from failures, from learning about the techniques and tools available and under development, and from understanding when, where and how they work best. Future management approaches will need to be taken in the context of climate change impacts on biodiversity, which will be a greater challenge.

Globally, it may not be possible to address all impacts from undesirable invasive alien species, but the successes described in this chapter show how management of biological invasions can deliver positive outcomes. This may save many threatened and endangered species and communities and improve ecosystems, nature's contributions to people and good quality of life, using the tools and approaches that have been and continue to be developed. Management of biological invasions is critical to improve ecosystem resilience and protect biodiversity in the context of future environmental changes, especially climate change.

REFERENCES

ABARES. (2021). *The National Priority List of Exotic Environmental Pests, Weeds and Diseases: Information Paper (Version 2.0)* (ABARES Report to Client Prepared for the Chief Environmental Biosecurity Officer). Department of Agriculture, Water and the Environment. https://www.agriculture.gov.au/sites/default/files/documents/eepl-information-paper.pdf

Abbott, C., Coulson, M., Gagné, N., Lacoursière-Roussel, A., Parent, G. J., Bajno, R., Dietrich, C., & May-McNally, S. (2021). *Guidance on the Use of Targeted Environmental DNA (eDNA) Analysis for the Management of Aquatic Invasive Species and Species at Risk: Vol. Research Document 2021/019*. Canadian Science Advisory Secretariat (CSAS) Ottawa, ON, Canada. https://waves-vagues.dfo-mpo.gc.ca/Library/40960791.pdf

Abbott, I. (2002). Origin and Spread of the cat, *Felis catus*, on mainland Australia, with a discussion of the magnitude of its early impact on native fauna. *Wildlife Research*, *29*(1), 51–74. https://doi.org/10.1071/WR01011

Aceves-Bueno, E., Adeleye, A. S., Feraud, M., Huang, Y., Tao, M., Yang, Y., & Anderson, S. E. (2017). The Accuracy of Citizen Science Data: A Quantitative Review. *Bulletin of the Ecological Society of America*, *98*(4), 278–290. https://www.jstor.org/stable/90013289

ACRDP. (2010). *Containment and Mitigation of Nuisance Tunicates on Prince Edward Island to Improve Mussel Farm Productivity* (Fact Sheet 6). Aquaculture Collaborative Research and Development Program. https://waves-vagues.dfo-mpo.gc.ca/library-bibliotheque/346732.pdf

Adams, C. I. M., Knapp, M., Gemmell, N. J., Jeunen, G.-J., Bunce, M., Lamare, M. D., & Taylor, H. R. (2019). Beyond Biodiversity: Can Environmental DNA (eDNA) Cut It as a Population Genetics Tool? *Genes*, *10*(3), 192. https://doi.org/10.3390/genes10030192

Adams, S. N. (1975). Sheep and Cattle Grazing in Forests: A Review. *Journal of Applied Ecology*, *12*(1), 143–152. https://doi.org/10.2307/2401724

Adams, V. M., & Setterfield, S. A. (2015). Optimal dynamic control of invasions: Applying a systematic conservation approach. *Ecological Applications*, *25*(4), 1131–1141. https://doi.org/10.1890/14-1062.1

Adolfi, A., Gantz, V. M., Jasinskiene, N., Lee, H.-F., Hwang, K., Terradas, G., Bulger, E. A., Ramaiah, A., Bennett, J. B., Emerson, J. J., Marshall, J. M., Bier, E., & James, A. A. (2020). Efficient population modification gene-drive rescue system in the malaria mosquito *Anopheles stephensi*. *Nature Communications*, *11*(1), 5553. https://doi.org/10.1038/s41467-020-19426-0

Adriaens, T., Branquart, E., Gosse, D., Reniers, J., & Vanderhoeven, S. (2019). *Feasibility of eradication and spread limitation for species of Union concern sensu the EU IAS Regulation (EU 1143/2014) in Belgium.* [Report prepared in support of implementing the IAS Regulation in Belgium]. Instituut voor Natuur- en Bosonderzoek. https://doi.org/10.21436/17033333

Adriaens, T., Vandegehuchte, M., & Casaer, J. (2018). *Guidance for drafting best management practices for invasive alien species* (Rapporten van Het Instituut Voor Natuur- En Bosonderzoek Nr. 68). Instituut voor Natuur- en Bosonderzoek. https://doi.org/10.21436/inbor.14912489

Agathokleous, E., Feng, Z., Iavicoli, I., & Calabrese, E. J. (2020). Nano-pesticides: A great challenge for biodiversity? The need for a broader perspective. *Nano Today*, *30*, 100808. https://doi.org/10.1016/j.nantod.2019.100808

Ågren, J. A., & Clark, A. G. (2018). Selfish genetic elements. *PLOS Genetics*, *14*(11), e1007700. https://doi.org/10.1371/journal.pgen.1007700

Aguanno, R. D., Lamielle, G., von Dobschuetz, S., & Dhingra, M. (2019). The power of evaluations: One year review of FAO's surveillance evaluation tool (SET) and tracking future impacts. *International Journal of Infectious Diseases*, *79*, 110–111. https://doi.org/10.1016/j.ijid.2018.11.274

Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Hernández-Montoya, J., Latofski-Robles, M., Luna-Mendoza, L., Méndez-Sánchez, F., Ortiz-Alcaraz, A., Rojas-Mayoral, E., & Samaniego-Herrera, A. (2018). The Conservation and Restoration of the Mexican Islands, a Successful Comprehensive and Collaborative Approach Relevant for Global Biodiversity. In A. Ortega-Rubio (Ed.), *Mexican Natural Resources Management and Biodiversity Conservation: Recent Case Studies* (pp. 177–192). Springer International Publishing. https://doi.org/10.1007/978-3-319-90584-6_9

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Méndez Sánchez, F., & Hernández-Montoya, J. C. (2016). La restauración ambiental exitosa de las islas de México: Una reflexión sobre los avances a la fecha y los retos por venir. In E. Ceccon & C. Martínez-Garza (Eds.), *Experiencias mexicanas en la restauración de los ecosistemas* (pp. 487–512). UNAM, UAEM, CONABIO. https://doi.org/10.22201/crim.9786070294778e.2017

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodriguez-Malagón, M., Félix-Lizárraga, M., Méndez Sánchez, F., González-Gómez, R., Torres-García, F., Hernández-Montoya, J. C., Barredo-Barberena, J. M., & Latofski-Robles, M. (2011). Eradications of invasive mammals on islands in Mexico: The roles of history and the collaboration between government agencies, local communities and a non-government organisation. In C. R. Veitch, M. N. Clout, & D. R. Towns (Eds.), *Island invasives: Eradication and management: Proceedings of the International Conference on Island Invasives* (pp. 386–394). IUCN and The Centre for Biodiversity and Biosecurity (CBB). https://www.academia.edu/50362808/Eradications_of_invasive_mammals_on_islands_in_Mexico_the_roles_of_history_and_the_collaboration_between_government_agencies_local_communities_and_a_non_government_organisation

Aiken, C., Davey, N., Chungul, K., Bach, T., Rangan, H., & Kull, C. (2015). *Managing Weeds on Bunuba Country in the Kimberley, Western Australia*. https://christiankull.files.wordpress.com/2016/01/aiken-et-al-2015-bunuba-weeds.pdf

Airoldi, L., & Bulleri, F. (2011). Anthropogenic Disturbance Can Determine the Magnitude of Opportunistic Species Responses on Marine Urban Infrastructures. *PLoS ONE*, *6*(8), e22985. https://doi.org/10.1371/journal.pone.0022985

Akter, S., Kompas, T., & Ward, M. B. (2015). Application of portfolio theory to asset-based biosecurity decision analysis. *Ecological Economics*, *117*, 73–85. https://doi.org/10.1016/j.ecolecon.2015.06.020

Alaniz, A. J., Núñez-Hidalgo, I., Carvajal, M. A., Alvarenga, T. M., Gómez-Cantillana, P., & Vergara, P. M. (2020). Current and future spatial assessment of biological control as a mechanism to reduce economic losses and carbon emissions: The case of *Solanum sisymbriifolium* in Africa. *Pest Management Science*, *76*(7), 2395–2405. https://doi.org/10.1002/ps.5776

Aldridge, D. C., Aldridge, S. L., Mead, A., Scales, H., Smith, R. K., Zieritz, A., & Sutherland, W. J. (2015). *Control of freshwater invasive species: Global evidence for the effects of selected interventions*. The University of Cambridge. http://dx.doi.org/10.13140/RG.2.2.22197.58084

Aldridge, D. C., Ho, S., & Froufe, E. (2014). The Ponto-Caspian quagga mussel, *Dreissena rostriformis bugensis* (Andrusov, 1897), invades Great Britain. *Aquatic Invasions*, *9*(4), 529–535. https://doi.org/10.3391/ai.2014.9.4.11

Alexander, J. M., Frankel, S. J., Hapner, N., Phillips, J. L., & Dupuis, V. (2017). Working across Cultures to Protect Native American Natural and Cultural Resources from Invasive Species in California. *Journal of Forestry*, *115*(5), 473–479. https://doi.org/10.5849/jof.16-018

Allan, B. F., Dutra, H. P., Goessling, L. S., Barnett, K., Chase, J. M., Marquis, R. J., Pang, G., Storch, G. A., Thach, R. E., & Orrock, J. L. (2010). Invasive honeysuckle eradication reduces tick-borne disease risk by altering host dynamics. *Proceedings of the National Academy of Sciences*, *107*(43), 18523–18527. https://doi.org/10.1073/pnas.1008362107

Amer, W., Ashong, S., & Tiomoko, D. (2015). *Management Manual for UNESCO Biosphere Reserves in Africa* (German Commission for UNESCO, Ed.). https://www.unesco.de/media/776

Amstrup, S. C., York, G., McDonald, T. L., Nielson, R., & Simac, K. (2004). Detecting Denning Polar Bears with Forward-Looking Infrared (FLIR) Imagery. *BioScience*, *54*(4), 337–344. https://doi.org/10.1641/0006-3568(2004)054[0337:DDPBWF]2.0.CO;2

Anandhi, S., Saminathan, V. R., Yasotha, P., Saravanan, P. T., & Rajanbabu, V. (2020). Nano-pesticides in pest management. *Journal of Entomology and Zoology Studies*, *8*(4), 685–690. https://www.entomoljournal.com/archives/2020/vol8issue4/PartL/8-3-380-752.pdf

Anderson, L. G., Chapman, J. K., Escontrela, D., & Gough, C. L. A. (2017). The role of conservation volunteers in the detection, monitoring and management of invasive alien lionfish. *Management of Biological Invasions*, *8*(4), 589–598. https://doi.org/10.3391/mbi.2017.8.4.14

Anderson, L. G., Rocliffe, S., Haddaway, N. R., & Dunn, A. M. (2015). The Role of Tourism and Recreation in the Spread of Non-Native Species: A Systematic Review and Meta-Analysis. *PLoS ONE*, *10*(10), e0140833. https://doi.org/10.1371/journal.pone.0140833

Anderson, L. W. J. (2005). California's reaction to *Caulerpa taxifolia*: A model for invasive species rapid response. *Biological Invasions*, *7*(6), 1003–1016. https://doi.org/10.1007/s10530-004-3123-z

Andreu, J., & Vilà, M. (2010). Risk analysis of potential invasive plants in Spain. *Journal for Nature Conservation*, *18*(1), 34–44. https://doi.org/10.1016/j.jnc.2009.02.002

Angujo, B. (2015, April 28). *Ugandan cultural King arbitrates Kenyan politician*. The Standard. https://www.standardmedia.co.ke/ureport/article/2000160058/ugandan-cultural-king-arbitrates-kenyan-politician

Anitha, V., Santheep, K. V., & Jyotsna Krishnakumar. (2017). Social and economic implications of *Mikania micrantha* in the Kerala Western Ghats. In C. A. Ellison, K. V. Sankaran, & S. T. Murphy (Eds.), *Invasive alien plants: Impacts on development and options for management* (pp. 29–38). CABI. https://doi.org/10.1079/9781780646275.0029

Anon. (2001). *Red Imported Fire Ants – Auckland Airport 2001*. https://www.cbd.int/doc/submissions/ias/ias-nz-ant-2007-en.pdf

Aquiloni, L., Becciolini, A., Berti, R., Porciani, S., Trunfio, C., & Gherardi, F. (2009). Managing invasive crayfish: Use of X-ray sterilisation of males. *Freshwater Biology*, *54*(7), 1510–1519. https://doi.org/10.1111/j.1365-2427.2009.02169.x

Argentine Naval Prefecture. (2021). *Good practices manual: Maintenance of vessels, equipment and infrastructure related to the different maritime activities*. https://www.argentina.gob.ar/sites/default/files/2023/06/manual_buenas_practicas_2020_0.pdf

Arndt, E., Robinson, A., & Hester, S. (2021). *Factors that influence vessel biofouling and its prevention and management* (p. 106) [Final report for CEBRA Project 190803]. https://cebra.unimelb.edu.au/__data/assets/pdf_file/0010/3822922/Endorsed-CEBRA-190803-Final-Report.pdf

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M., Mörtberg, U., … Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, *281*(1780), 20133330. https://doi.org/10.1098/rspb.2013.3330

Arora, D. K. (Ed.). (2003). *Fungal Biotechnology in Agricultural, Food, and Environmental Applications*. CRC Press. https://doi.org/10.1201/9780203913369

Asa, C., & Moresco, A. (2019). Fertility Control in Wildlife: Review of Current Status, Including Novel and Future Technologies. In P. Comizzoli, J. L. Brown, & W. V. Holt (Eds.), *Reproductive Sciences in Animal Conservation* (Vol. 1200, pp. 507–543). Springer International Publishing. https://doi.org/10.1007/978-3-030-23633-5_17

Asare-Nuamah, P. (2022). Smallholder farmers' adaptation strategies for the management of fall armyworm (*Spodoptera frugiperda*) in rural Ghana. *International Journal of Pest Management*, *68*(1), 8–18. https://doi.org/10.1080/09670874.2020.1787552

Ascensão, F., & Capinha, C. (2017). Aliens on the Move: Transportation Networks and Non-native Species. In L. Borda-de-Água, R. Barrientos, P. Beja, & H. M. Pereira (Eds.), *Railway Ecology* (pp. 65–80). Springer International Publishing. https://doi.org/10.1007/978-3-319-57496-7_5

Atalah, J., Brook, R., Cahill, P., Fletcher, L. M., & Hopkins, G. A. (2016). It's a wrap: Encapsulation as a management tool for marine biofouling. *Biofouling*, *32*(3), 277–286. https://doi.org/10.1080/08927014.2015.1137288

Atalah, J., Hopkins, G. A., Fletcher, L. M., Castinel, A., & Forrest, B. M. (2015). Concepts for biocontrol in marine environments: Is there a way forward? *Management of Biological Invasions*, *6*(1), 1–12. https://doi.org/10.3391/mbi.2015.6.1.01

Atkinson, I., & Sheppard, A. (2000). Brooms as part of the Australian nursery industry. *Plant Protection Quarterly*, *15*(4), 176–178. https://caws.org.nz/PPQ131415/PPQ%2015-4%20pp176-178%20Atkinson.pdf

Atyosi, Z., Ramarumo, L. J., & Maroyi, A. (2019). Alien Plants in the Eastern Cape Province in South Africa: Perceptions of Their Contributions to Livelihoods of Local Communities. *Sustainability*, *11*(18), 5043. https://doi.org/10.3390/su11185043

Augustin, S., Boonham, N., De Kogel, W. J., Donner, P., Faccoli, M., Lees, D. C., Marini, L., Mori, N., Petrucco Toffolo, E., Quilici, S., Roques, A., Yart, A., & Battisti, A. (2012). A review of pest surveillance techniques for detecting quarantine pests in Europe. *EPPO Bulletin*, *42*(3), 515–551. https://doi.org/10.1111/epp.2600

Auld, B., & Johnson, S. B. (2014). Invasive alien plant management. *CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources*, *9*(037). https://doi.org/10.1079/PAVSNNR20149037

Australian Academy of Science. (2017). *Synthetic gene drives in Australia: Implications of emerging technologies*. https://www.science.org.au/support/analysis/reports/synthetic-gene-drives-australia-implications-emerging-technologies

Australian Department of Agriculture. (2019). *Final pest risk analysis for brown marmorated stink bug (*Halyomorpha halys*)* (CC BY 3.0). https://www.awe.gov.au/sites/default/files/documents/final-bmsb-pra-report.pdf

Australian Government. (2021a). *Innovative Biosecurity 3D X-ray Project—Department of agriculture, fisheries and forestry*. Biosecurity and Trade in Australia. https://www.agriculture.gov.au/biosecurity-trade/policy/australia/biosecurity-3d-x-ray

Australian Government. (2021b). *Seasonal measures for Brown marmorated stink bug (BMSB)—Department of Agriculture, Fisheries and Forestry*. Biosecurity and Trade in Australia. https://www.agriculture.gov.au/biosecurity-trade/import/before/brown-marmorated-stink-bugs

Bach, T. M., Kull, C. A., & Rangan, H. (2019). From killing lists to healthy country: Aboriginal approaches to weed control in the Kimberley, Western Australia. *Journal of Environmental Management*, *229*, 182–192. https://doi.org/10.1016/j.jenvman.2018.06.050

Bacher, S., Blackburn, T. M., Essl, F., Genovesi, P., Heikkilä, J., Jeschke, J. M., Jones, G., Keller, R., Kenis, M., Kueffer, C., Martinou, A. F., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Richardson, D. M., Roy, H. E., Saul, W.-C., Scalera, R., … Kumschick, S. (2018). Socio-economic impact classification of alien taxa (SEICAT). *Methods in Ecology and Evolution*, *9*(1), 159–168. https://doi.org/10.1111/2041-210X.12844

Bacon, S. J., Bacher, S., & Aebi, A. (2012). Gaps in Border Controls Are Related to Quarantine Alien Insect Invasions in Europe. *PLoS ONE*, *7*(10), e47689. https://doi.org/10.1371/journal.pone.0047689

Bailey, S. A. (2015). An overview of thirty years of research on ballast water as a vector for aquatic invasive species to freshwater and marine environments. *Aquatic Ecosystem Health & Management*, *18*(3), 261–268. https://doi.org/10.1080/14634988.2015.1027129

Bailey, S. A., Brown, L., Campbell, M. L., Canning-Clode, J., Carlton, J. T., Castro, N., Chainho, P., Chan, F. T., Creed, J. C., Curd, A., Darling, J., Fofonoff, P., Galil, B. S., Hewitt, C. L., Inglis, G. J., Keith, I., Mandrak, N. E., Marchini, A., McKenzie, C. H., … Zhan, A. (2020). Trends in the detection of aquatic non-indigenous species across global marine, estuarine and freshwater ecosystems: A 50-year perspective. *Diversity and Distributions*, *26*(12), 1780–1797. https://doi.org/10.1111/ddi.13167

Bailey, S. A., Deneau, M. G., Jean, L., Wiley, C. J., Leung, B., & MacIsaac, H. J. (2011). Evaluating Efficacy of an Environmental Policy to Prevent Biological Invasions. *Environmental Science & Technology*, *45*(7), 2554–2561. https://doi.org/10.1021/es102655j

Bajer, P. G., Chizinski, C. J., & Sorensen, P. W. (2011). Using the Judas technique to locate and remove wintertime aggregations of invasive common carp. *Fisheries Management and Ecology*, *18*(6), 497–505. https://doi.org/10.1111/j.1365-2400.2011.00805.x

Bajwa, A. A., Farooq, M., Nawaz, A., Yadav, L., Chauhan, B. S., & Adkins, S. (2019). Impact of invasive plant species on the livelihoods of farming households: Evidence from *Parthenium hysterophorus* invasion in rural Punjab, Pakistan. *Biological Invasions*, *21*(11), 3285–3304. https://doi.org/10.1007/s10530-019-02047-0

Baker, A. C., Glynn, P. W., & Riegl, B. (2008). Climate Change and Coral Reef Bleaching: An Ecological Assessment of Long-Term Impacts, Recovery Trends and Future Outlook. *Estuarine Coastal and Shelf Science*, *80*(4), 435–471. https://doi.org/10.1016/j.ecss.2008.09.003

Ball, D., Ross, P., English, A., Patten, T., Upcroft, B., Fitch, R., Sukkarieh, S., Wyeth, G., & Corke, P. (2015). Robotics for Sustainable Broad-Acre Agriculture. In L. Mejias, P. Corke, & J. Roberts (Eds.), *Field and Service Robotics: Results of the 9th International Conference* (Vol. 105, pp. 439–453). Springer International Publishing. https://doi.org/10.1007/978-3-319-07488-7_30

Ballari, S. A., Cuevas, M. F., Ojeda, R. A., & Navarro, J. L. (2015). Diet of wild boar (*Sus scrofa*) in a protected area of Argentina: The importance of baiting. *Mammal Research*, *60*(1), 81–87. https://doi.org/10.1007/s13364-014-0202-0

Banchi, E., Ametrano, C. G., Stanković, D., Verardo, P., Moretti, O., Gabrielli, F., Lazzarin, S., Borney, M. F., Tassan, F., Tretiach, M., Pallavicini, A., & Muggia, L. (2018). DNA metabarcoding uncovers fungal diversity of mixed airborne samples in Italy. *PLoS ONE*, *13*(3), e0194489. https://doi.org/10.1371/journal.pone.0194489

Banerjee, A. K., Khuroob, A. A., Dehnen-Schmutz, K., Pant, V., Patwardhan, C., Bhowmick, A. R., & Mukherjee, A. (2021). An integrated policy framework and plan of action to prevent and control plant invasions in India. *Environmental Science and Policy*, *124*, 64–72. https://doi.org/10.1016/j.envsci.2021.06.003

Banks, N. C., Paini, D. R., Bayliss, K. L., & Hodda, M. (2015). The role of global trade and transport network topology in the human-mediated dispersal of alien species. *Ecology Letters*, *18*(2), 188–199. https://doi.org/10.1111/ele.12397

Barasa, M. W. (2012). *Celebrating Bukusu tradition at the cultural Festival*. Monitor. https://www.monitor.co.ug/uganda/lifestyle/reviews-profiles/celebrating-bukusu-tradition-at-the-cultural-festival-1507172

Barber, D., & Glass, P. (2015). Yellomundee Firesticks. In E. J. Ens, J. Fisher, & O. Costello (Eds.), *Indigenous people and invasive species: Perceptions, management, challenges and uses*. IUCN Commission on Ecosystem Management Community Report. https://ipm.ifas.ufl.edu/pdfs/ens_et_al_2015_indigenous_people_and_invasive_species_iucn_cem_ecosystems_and_invasiv.pdf

Barbet-Massin, M., Rome, Q., Villemant, C., & Courchamp, F. (2018). Can species distribution models really predict the expansion of invasive species? *PLoS ONE*, *13*(3), e0193085. https://doi.org/10.1371/journal.pone.0193085

Bardal, H. (2019). Small- and large-scale eradication of invasive fish and fish parasites in freshwater systems in Norway. In C. R. Veitch, M. N. Clout, Martin, A.R., Russel, J.C., & West, C.J. (Eds.), *Island invasives: Scaling up to meet the challenge* (pp. 447–451). IUCN.

Barham, E., Sharrock, S., Lane, C., & Baker, R. (2016). The International Plant Sentinel Network: A tool for Regional and National Plant Protection Organizations. *EPPO Bulletin*, *46*(1), 156–162. https://doi.org/10.1111/epp.12283

Bariw, S. A., Kudadze, S., & Adzawla, W. (2020). Prevalence, effects and management of fall army worm in the Nkoranza South Municipality, Bono East region of Ghana. *Cogent Food & Agriculture*, *6*(1), 1800239. https://doi.org/10.1080/23311932.2020.1800239

Barratt, B. I. P., Colmenarez, Y. C., Day, M. D., Ivey, P., Klapwijk, J. N., Loomans, A. J. M., Mason, P. G., Palmer, W. A., Sankaran, K. V., & Zhang, F. (2021). Regulatory challenges for biological control. In P. G. Mason (Ed.), *Biological Control: Global Impacts, Challenges and Future Directions of Pest Management* (pp. 166–196). CSIRO. https://ebooks.publish.csiro.au/content/biological-control

Bartula, M., Stojšić, V., Perić, R., & Kitnæs, K. S. (2011). Protection of Natura 2000 Habitat Types in the Ramsar Site "Zasavica Special Nature Reserve" in Serbia. *Natural Areas Journal*, *31*(4), 349–357. https://doi.org/10.3375/043.031.0405

Batista, P. M., Andreotti, R., Chiang, J. O., Ferreira, M. S., & Vasconcelos, P. F. da C. (2012). Seroepidemiological monitoring in sentinel animals and vectors as part of arbovirus surveillance in the state of Mato Grosso do Sul, Brazil. *Revista Da Sociedade Brasileira de Medicina Tropical*, *45*(2), 168–173. https://doi.org/10.1590/S0037-86822012000200006

Baum, J. A., Bogaert, T., Clinton, W., Heck, G. R., Feldmann, P., Ilagan, O., Johnson, S., Plaetinck, G., Munyikwa, T., Pleau, M., Vaughn, T., & Roberts, J. (2007). Control of coleopteran insect pests through RNA interference. *Nature Biotechnology*, *25*(11), Article 11. https://doi.org/10.1038/nbt1359

Bawden, O., Kulk, J., Russell, R., McCool, C., English, A., Dayoub, F., Lehnert, C., & Perez, T. (2017). Robot for weed species plant-specific management. *Journal of Field Robotics*, *34*(6), 1179–1199. https://doi.org/10.1002/rob.21727

Bax, N. J., & Thresher, R. E. (2009). Ecological, behavioral, and genetic factors influencing the recombinant control of invasive pests. *Ecological Applications*, *19*(4), 873–888. https://doi.org/10.1890/07-1588.1

Bax, N. J., Williamson, A., Aguero, M., Gonzalez, E., & Geeves, W. (2003). Marine invasive alien species: A threat to global biodiversity. *Marine Policy*, *27*(4), 313–323. https://doi.org/10.1016/S0308-597X(03)00041-1

Beauchamp, A. J., & Ward, E. (2011). A targeted approach to multi-species control and eradication of escaped garden and ecosystem modifying weeds on Motuopao Island, Northland, New Zealand. In C. R. Veitch, M. N. Clout, & D. R. Towns (Eds.), *Island Invasives: Eradication and Management* (pp. 264–268). IUCN and The Centre for Biodiversity and Biosecurity (CBB). https://portals.iucn.org/library/efiles/documents/ssc-op-042.pdf

Beaumont, L. J., Gallagher, R. V., Leishman, M. R., Hughes, L., & Downey, P. O. (2014). How can knowledge of the climate niche inform the weed risk assessment process? A case study of *Chrysanthemoides monilifera* in Australia. *Diversity and Distributions*, *20*(6), 613–625. https://doi.org/10.1111/ddi.12190

Beaumont, L. J., Gallagher, R. V., Thuiller, W., Downey, P. O., Leishman, M. R., & Hughes, L. (2009). Different climatic envelopes among invasive populations may lead to underestimations of current and future biological invasions. *Diversity and Distributions*, *15*(3), 409–420. https://doi.org/10.1111/j.1472-4642.2008.00547.x

Beaury, E. M., Fusco, E. J., Jackson, M. R., Laginhas, B. B., Morelli, T. L., Allen, J. M., Pasquarella, V. J., & Bradley, B. A. (2020). Incorporating climate change into invasive species management: Insights from managers. *Biological Invasions*, *22*(2), 233–252. https://doi.org/10.1007/s10530-019-02087-6

Beckett, S. (2008). *Simulation modelling environment for pandemic influenza*.

Beckett, S., & Garner, M. G. (2007). Simulating disease spread within a geographic information system environment. *Veterinaria Italiana*, *43*(3), 595–604. https://pubmed.ncbi.nlm.nih.gov/20422538/

Beckie, H. J., Ashworth, M. B., & Flower, K. C. (2019). Herbicide Resistance Management: Recent Developments and Trends. *Plants*, *8*(6), 161. https://doi.org/10.3390/plants8060161

Bedolla-Guzmán, Y., Méndez-Sánchez, F., Aguirre-Muñoz, A., Félix-Lizárraga, M., Fabila-Blanco, A., Bravo-Hernández, E., Hernández-Ríos, A., Corrales-Sauceda, M., Aguilar-Vargas, A., Aztorga-Ornelas, A., Solís-Carlos, F., Torres-García, F., Luna-Mendoza, L., Ortiz-Alcaraz, A., Hernández-Montoya, J., Latofski-Robles, M., Rojas-Mayoral, E., & Cárdenas-Tapia, A. (2019). Recovery and current status of seabirds on the Baja California Pacific Islands, Mexico, following restoration actions. In C. R. Veitch, M. N. Clout, A. R. Martin, J. C. Russell, & C. J. West (Eds.), *Island Invasives: Scaling up to meet the challenge. Proceedings of the international conference on island invasives 2017* (pp. 531-538.). IUCN. https://library.sprep.org/sites/default/files/2021-07/recovery-current-status-seabirds.pdf

Beever, E. A., Simberloff, D., Crowley, S. L., Al-Chokhachy, R., Jackson, H. A., & Petersen, S. L. (2019). Social–ecological mismatches create conservation challenges in introduced species management. *Frontiers in Ecology and the Environment*, *17*(2), 117–125. https://doi.org/10.1002/fee.2000

Begley, C., Rainbow, R., & Younus, F. (2021). *Feral Futures: Overview of technology opportunities for vertebrate pest management*. Spiegare Pty Limited. Published by the Centre for Invasive Species Solutions, Canberra, Australia. https://invasives.com.au/wp-content/uploads/2021/05/Feral-Futures-Overview-of-Technology-Opportunities-web.pdf

Bejakovich, D., Bingham, K., Butland, B., Greenaway, A., Harrison, A., Hastie, K., Hodges, D., MacDonald, E., Mark-Shadbolt, M., Marsh, A., Massey, E., O'Brien, C., Palmer, R., Ryan, J., Walsh, J., & Whitworth, M. (2018). *Biosecurtity 2025—Engagement plan. Strategic Direction 1.* (p. 43). https://www.thisisus.nz/assets/Resources/a6783d1f81/Engagement-plan-Strategic-Direction-1.pdf

Bengsen, A. J., Butler, J. A., Masters, P., Bengsen, A. J., Butler, J. A., & Masters, P. (2012). Applying home-range and landscape-use data to design effective feral-cat control programs. *Wildlife Research*, *39*(3), 258–265. https://doi.org/10.1071/WR11097

Bengsen, A. J., & Sparkes, J. (2016). Can recreational hunting contribute to pest mammal control on public land in Australia? *Mammal Review*, *46*(4), 297–310. https://doi.org/10.1111/mam.12070

Benhalima, H., Chaudhry, M. Q., Mills, K. A., & Price, N. R. (2004). Phosphine resistance in stored-product insects collected from various grain storage facilities in Morocco. *Journal of Stored Products Research*, *40*(3), 241–249. https://doi.org/10.1016/S0022-474X(03)00012-2

Bennett, J. M. (2015). Agricultural big data: Utilisation to discover the unknown and instigate practice change. *Farm Policy Journal*, *12*(1), 43–50. http://www.farminstitute.org.au/publications-1/farm-policy-journals/2015-autumn-from-little-data-big-data-grow/agricultural-big-data-utilisation-to-discover-the-unknown-and-instigate-practice-change

Berna, A. Z., Anderson, A. R., & Trowell, S. C. (2009). Bio-Benchmarking of Electronic Nose Sensors. *PLoS ONE*, *4*(7), e6406. https://doi.org/10.1371/journal.pone.0006406

Bertolino, S., & Genovesi, P. (2003). Spread and attempted eradication of the grey squirrel (*Sciurus carolinensis*) in Italy, and consequences for the red squirrel (*Sciurus vulgaris*) in Eurasia. *Biological Conservation*, *109*(3), 351–358. https://doi.org/10.1016/S0006-3207(02)00161-1

Besacier-Monbertrand, A.-L., Paillex, A., & Castella, E. (2014). Short-Term Impacts of Lateral Hydrological Connectivity Restoration on Aquatic Macroinvertebrates. *River Research and Applications*, *30*(5), 557–570. https://doi.org/10.1002/rra.2597

Bester, M. N., Bloomer, J. P., Van Aarde, R. J., Erasmus, B. H., Van Rensburg, P. J. J., Skinner, J. D., Howell, P. G., & Naude, T. W. (2002). A review of the successful eradication of feral cats from sub-Antarctic Marion Island, Southern Indian Ocean. *South African Journal of Wildlife Research – 24-Month Delayed Open Access*, *32*(1), 65–73. https://journals.co.za/content/wild/32/1/EJC117137

Bigler, F., Babendreier, D., & Kuhlmann, U. (Eds.). (2006). *Environmental Impact of Invertebrates for Biological Control of Arthropods: Methods and Risk Assessment*. CABI Publishing. http://sherekashmir.informaticspublishing.com/315/1/9780851990583.pdf

Binggeli, P. (2003). Cactaceae, *Opuntia* spp., prickly pear, raiketa, rakaita, raketa. In S. M. Goodman & J. P. Benstead (Eds.), *The natural history of Madagascar* (pp. 335–339). University of Chicago Press. https://www.researchgate.net/publication/278848886

Bingham, P., Brangenberg, N., Williams, R., & Van Andel, M. (2013). Investigation into the first diagnosis of Ostreid Herpesvirus Type 1 in Pacific oysters. *Surveillance*, *40*(2), 20–24. https://www.academia.edu/75251990/Investigation_Into_the_First_Diagnosis_of_Ostreid_Herpesvirus_Type_1_in_Pacific_Oysters

Binns, J. A., Illgner, P. M., & Nel, E. L. (2001). Water shortage, deforestation and development: South Africa's working for water programme. *Land Degradation & Development*, *12*(4), 341–355. https://doi.org/10.1002/ldr.455

Birand, A., Cassey, P., Ross, J. V., Russell, J. C., Thomas, P., & Prowse, T. A. A. (2022). Gene drives for vertebrate pest control: Realistic spatial modelling of eradication probabilities and times for island mouse populations. *Molecular Ecology*, *31*(6), 1907–1923. https://doi.org/10.1111/mec.16361

Bithas, K., Latinopoulos, D., Kolimenakis, A., & Richardson, C. (2018). Social Benefits from Controlling Invasive Asian Tiger and Native Mosquitoes: A Stated Preference Study in Athens, Greece. *Ecological Economics*, *145*, 46–56. https://doi.org/10.1016/j.ecolecon.2017.08.017

Black, R., & Bartlett, D. M. F. (2020). Biosecurity frameworks for cross-border movement of invasive alien species. *Environmental Science & Policy*, *105*, 113–119. https://doi.org/10.1016/j.envsci.2019.12.011

Bleach, C. (2019). Plant Health Surveillance and Incursion Investigation Annual report. *Surveillance (Wellington)*, *46*(3), 71–80. https://www.mpi.govt.nz/dmsdocument/37564-Surveillance-Magazine-Vol-46-No-3-September-2019-includes-Annual-Report

Bomford, M., & Hart, Q. (1999). Assessing the Risk Associated with Importing and Keeping Exotic Vertebrates in Australia. *The Australian Journal of Emergency Management*, *14*(3), 16–19. https://search.informit.com.au/documentSummary;dn=391999856543177;res=IELAPA

Bomford, M., & O'Brien, P. (1995). Eradication or Control for Vertebrate Pests? *Wildlife Society Bulletin (1973-2006)*, *23*(2), 249–255. JSTOR. https://www.jstor.org/stable/3782799

Booth, C. (2010). Hunting & feral animal control: Conservation or con? In M. Tensen & B. Jones (Eds.), *Proceedings of the RSPCA Australia Scientific Seminar Convergence or conflict: Animal welfare in wildlife management and conservation* (p. 5). Invasive Species Council. https://www.rspca.org.au/sites/default/files/website/The-facts/Science/Scientific-Seminar/2010/SciSem2010-Proceedings.pdf#page=27

Booth, E. D., Rawlinson, P. J., Maria Fagundes, P., & Leiner, K. A. (2017). Regulatory requirements for genotoxicity assessment of plant protection product active ingredients, impurities, and metabolites. *Environmental and Molecular Mutagenesis*, *58*(5), 325–344. https://doi.org/10.1002/em.22084

Booy, O. (2019). *Prioritising the management of invasive non-native species* [Ph.D. Thesis, Newcastle University, UK.]. https://theses.ncl.ac.uk/jspui/handle/10443/4926

Booy, O., Mill, A. C., Roy, H. E., Hiley, A., Moore, N., Robertson, P., Baker, S., Brazier, M., Bue, M., Bullock, R., Campbell, S., Eyre, D., Foster, J., Hatton-Ellis, M., Long, J., Macadam, C., Morrison-Bell, C., Mumford, J., Newman, J., … Wyn, G. (2017). Risk management to prioritise the eradication of new and emerging invasive non-native species. *Biological Invasions*, *19*(8), 2401–2417. https://doi.org/10.1007/s10530-017-1451-z

Booy, O., Robertson, P. A., Moore, N., Ward, J., Roy, H. E., Adriaens, T., Shaw, R., Van Valkenburg, J., Wyn, G., Bertolino, S., Blight, O., Branquart, E., Brundu, G., Caffrey, J., Capizzi, D., Casaer, J., De Clerck, O., Coughlan, N. E., Davis, E., … Mill, A. C. (2020). Using structured eradication feasibility assessment to prioritize the management of new and emerging invasive alien species in Europe. *Global Change Biology*, *26*(11), 6235–6250. https://doi.org/10.1111/gcb.15280

Bortolus, A. (2008). Error Cascades in the Biological Sciences: The Unwanted Consequences of Using Bad Taxonomy in Ecology. *AMBIO: A Journal of the Human Environment*, *37*(2), 114–118. https://doi.org/10.1579/0044-7447(2008)37[114:ECITBS]2.0.CO;2

Bottrill, M. C., Joseph, L. N., Carwardine, J., Bode, M., Cook, C., Game, E. T., Grantham, H., Kark, S., Linke, S., McDonald-Madden, E., Pressey, R. L., Walker, S., Wilson, K. A., & Possingham, H. P. (2008). Is conservation triage just smart decision making? *Trends in Ecology & Evolution*, *23*(12), 649–654. https://doi.org/10.1016/j.tree.2008.07.007

Bourdôt, G. W., Kriticos, D. J., & Dodd, M. B. (2018). A national weed management decision-support system. *21st Australasian Weeds Conference*, 82–85. https://caws.org.nz/old-site/awc/2018/awc201810821.pdf

Boy, G., & Witt, A. (2013). *Invasive alien plants and their management in Africa*. UNEP/GEF Removing Barriers to Invasive Plant Management Project International Coordination Unit. https://www.cabi.org/

Uploads/CABI/publishing/promotional-materials/african-invasives-book.pdf

Boyd, J., Epanchin-Niell, R., & Siikamaki, J. (2015). Conservation Planning: A Review of Return on Investment Analysis. *Review of Environmental Economics and Policy*, *9*(1), 23–42. https://doi.org/10.1093/reep/reu014

Bradhurst, R. A., Roche, S. E., East, I. J., Kwan, P., & Garner, M. G. (2015). A hybrid modeling approach to simulating foot-and-mouth disease outbreaks in Australian livestock. *Frontiers in Environmental Science*, *3*(17). https://doi.org/10.3389/fenvs.2015.00017

Bradie, J., & Leung, B. (2015). Pathway-level models to predict non-indigenous species establishment using propagule pressure, environmental tolerance and trait data. *Journal of Applied Ecology*, *52*(1), 100–109. https://doi.org/10.1111/1365-2664.12376

Brancalion, P. H. S., Campoe, O., Mendes, J. C. T., Noel, C., Moreira, G. G., van Melis, J., Stape, J. L., & Guillemot, J. (2019). Intensive silviculture enhances biomass accumulation and tree diversity recovery in tropical forest restoration. *Ecological Applications*, *29*(2), e01847. https://www.jstor.org/stable/26623324

Briese, D. T. (2000). Classical biological control. In B. M. Sindel (Ed.), *Australian Weed Management Systems* (pp. 161–192). R.G. and F.J. Richardson. https://www.cabdirect.org/cabdirect/abstract/20023012716

Brockerhoff, E. G., Liebhold, A. M., Richardson, B., & Suckling, D. M. (2010). Eradication of invasive forest insects: Concepts, methods, costs and benefits. *New Zealand Journal of Forestry Science*, *40*, S117–S135. https://www.fs.usda.gov/treesearch/pubs/34736

Broderstad, E. G., & Eythórsson, E. (2014). Resilient communities? Collapse and recovery of a social-ecological system in Arctic Norway. *Ecology and Society*, *19*(3), 1. https://doi.org/10.5751/ES-06533-190301

Brown, P. M. J., Adriaens, T., Bathon, H., Cuppen, J., Goldarazena, A., Hägg, T., Kenis, M., Klausnitzer, B. E. M., Kovář, I., Loomans, A. J. M., Majerus, M. E. N., Nedved, O., Pedersen, J., Rabitsch, W., Roy, H. E., Ternois, V., Zakharov, I. A., & Roy, D. B. (2008). *Harmonia axyridis* in Europe: Spread and distribution of a non-native coccinellid. In H. E. Roy & E. Wajnberg (Eds.), *From Biological Control to Invasion: The Ladybird Harmonia axyridis as a Model Species* (pp. 5–21). Springer Netherlands. https://doi.org/10.1007/978-1-4020-6939-0_2

Browne, C., Stafford, K., & Fordham, R. (2006). The use of scent-detection dogs. *Irish Veterinary Journal*, *59*(2), 97–104. https://www.researchgate.net/publication/261663456

Brunel, S., Branquart, E., Fried, G., Van Valkenburg, J., Brundu, G., Starfinger, U., Buholzer, S., Uludag, A., Joseffson, M., & Baker, R. (2010). The EPPO prioritization process for invasive alien plants. *EPPO Bulletin*, *40*(3), 407–422. https://doi.org/10.1111/j.1365-2338.2010.02423.x

Buchadas, A., Vaz, A. S., Honrado, J. P., Alagador, D., Bastos, R., Cabral, J. A., Santos, M., & Vicente, J. R. (2017). Dynamic models in research and management of biological invasions. *Journal of Environmental Management*, *196*, 594–606. https://doi.org/10.1016/j.jenvman.2017.03.060

Buckley, Y. M., Bolker, B. M., & Rees, M. (2007). Disturbance, invasion and re-invasion: Managing the weed-shaped hole in disturbed ecosystems. *Ecology Letters*, *10*(9), 809–817. https://doi.org/10.1111/j.1461-0248.2007.01067.x

Buckley, Y. M., Briese, D. T., & Rees, M. (2003a). Demography and management of the invasive plant species *Hypericum perforatum*. I. Using multi-level mixed-effects models for characterizing growth, survival and fecundity in a long-term data set. *Journal of Applied Ecology*, *40*(3), 481–493. https://doi.org/10.1046/j.1365-2664.2003.00821.x

Buckley, Y. M., Briese, D. T., & Rees, M. (2003b). Demography and management of the invasive plant species *Hypericum perforatum*. II. Construction and use of an individual-based model to predict population dynamics and the effects of management strategies. *Journal of Applied Ecology*, *40*(3), 494–507. https://doi.org/10.1046/j.1365-2664.2003.00822.x

Buckley, Y. M., Rees, M., Sheppard, A. W., & Smyth, M. J. (2005). Stable coexistence of an invasive plant and biocontrol agent: A parameterized coupled plant–herbivore model. *Journal of Applied Ecology*, *42*(1), 70–79. https://doi.org/10.1111/j.1365-2664.2005.00991.x

Bugnot, A. B., Mayer-Pinto, M., Airoldi, L., Heery, E. C., Johnston, E. L., Critchley, L. P., Strain, E. M. A., Morris, R. L., Loke, L. H. L., Bishop, M. J., Sheehan, E. V., Coleman, R. A., & Dafforn, K. A. (2021). Current and projected global extent of marine built structures. *Nature Sustainability*, *4*(1), 33–41. https://doi.org/10.1038/s41893-020-00595-1

Bulleri, F., & Airoldi, L. (2005). Artificial marine structures facilitate the spread of a non-indigenous green alga, *Codium fragile* ssp. *Tomentosoides*, in the north Adriatic Sea. *Journal of Applied Ecology*, *42*(6), 1063–1072. https://doi.org/10.1111/j.1365-2664.2005.01096.x

Burgiel, S., Foote, G., Orellana, M., & Perrault, A. (2006). *Invasive Alien Species and Trade: Integrating Prevention Measures and International Trade Rules* (p. 54). Center for International Environmental Law (CIEL) and Defenders of Wildlife. https://cleantrade.typepad.com/clean_trade/files/invasives_trade_paper_0106.pdf

Burgman, M. (2005). *Risks and Decisions for Conservation and Environmental Management*. Cambridge University Press. https://doi.org/10.1017/CBO9780511614279

Burke, G., Singh, B. R., & Theodore, L. (2005). *Handbook of environmental management and technology.* (Issue Ed. 2). John Wiley & Sons, Inc.

Burrows, N. D. (2018). Feral Animals in the Semi-arid and Arid Regions of Australia: Origins, Impacts and Control. In H. Lambers (Ed.), *On the Ecology of Australia's Arid Zone* (pp. 331–373). Springer International Publishing. https://doi.org/10.1007/978-3-319-93943-8_13

Burt, J. W., Muir, A. A., Piovia-Scott, J., Veblen, K. E., Chang, A. L., Grossman, J. D., & Weiskel, H. W. (2007). Preventing horticultural introductions of invasive plants: Potential efficacy of voluntary initiatives. *Biological Invasions*, *9*(8), 909–923. https://doi.org/10.1007/s10530-007-9090-4

Bylemans, J., Gleeson, D. M., Duncan, R. P., Hardy, C. M., & Furlan, E. M. (2019). A performance evaluation of targeted eDNA and eDNA metabarcoding analyses for freshwater fishes. *Environmental DNA*, *1*(4), 402–414. https://doi.org/10.1002/edn3.41

Byun, C., de Blois, S., & Brisson, J. (2013). Plant functional group identity and diversity determine biotic resistance to invasion by an exotic grass. *Journal of Ecology*, *101*(1), 128–139. https://doi.org/10.1111/1365-2745.12016

Byun, C., de Blois, S., & Brisson, J. (2018). Management of invasive plants through ecological resistance. *Biological Invasions*, *20*(1), 13–27. https://doi.org/10.1007/s10530-017-1529-7

Byun, C., de Blois, S., & Brisson, J. (2020). Restoring functionally diverse communities enhances invasion resistance in a freshwater wetland. *Journal of Ecology*, *108*(6), 2485–2498. https://doi.org/10.1111/1365-2745.13419

Byun, C., Oh, M., Lee, E. J., & Kang, H. (2020). Seed density is as important as limiting similarity, diversity effect, and propagule pressure in plant restoration to control invasion. *Ecological Engineering*, *144*, 105712. https://doi.org/10.1016/j.ecoleng.2019.105712

CABI. (2021). *Horizon Scanning Tool—Free version*. https://www.cabi.org/HorizonScanningTool/?auth=true

Caceres-Escobar, H., Kark, S., Atkinson, S. C., Possingham, H. P., & Davis, K. J. (2019). Integrating local knowledge to prioritise invasive species management. *People and Nature*, *1*(2), 220–233. https://doi.org/10.1002/pan3.27

Cacho, O. J., Spring, D., Hester, S., & Mac Nally, R. (2010). Allocating surveillance effort in the management of invasive species: A spatially-explicit model. *Environmental Modelling & Software*, *25*(4), 444–454. https://doi.org/10.1016/j.envsoft.2009.10.014

Cadenasso, M. L., & Pickett, S. T. A. (2008). Urban Principles for Ecological Landscape Design and Maintenance: Scientific Fundamentals. *Cities and the Environment (CATE)*, *1*(2), 4. https://digitalcommons.lmu.edu/cate/vol1/iss2/4

CADIC-CONICET. (2020, May 19). *Control de plagas en el predio del CADIC*. https://cadic.conicet.gov.ar/?post_type=post&p=4896

Calderón Alvarez, C., Causton, C. E., Hoddle, M. S., Hoddle, C. D., van Driesche, R., & Stanek, E. J. (2012). Monitoring the effects of *Rodolia cardinalis* on *Icerya purchasi* populations on the Galapagos Islands. *BioControl*, *57*(2), 167–179. https://doi.org/10.1007/s10526-011-9429-8

Calero, M. L., & Monti, G. (2022). Assessment of the Current Surveillance System for Human Leptospirosis in Ecuador by Decision Analytic Modeling. *Frontiers in Public Health*, *10*, 711938. https://doi.org/10.3389/fpubh.2022.711938

Calvete, C. (2006). Modeling the Effect of Population Dynamics on the Impact of Rabbit Hemorrhagic Disease. *Conservation Biology*, *20*(4), 1232–1241. https://doi.org/10.1111/j.1523-1739.2006.00371.x

Camac, J., Baumgartner, J., Garms, B., Robinson, A., & Kompas, T. (2021). *Estimating trading partner exposure risk to new pests or diseases* (p. 59) [Technical Report for CEBRA project 190606]. https://cebra.unimelb.edu.au/__data/assets/pdf_file/0006/3825834/190606_finalreport.pdf

Camac, J., Baumgartner, J., Robinson, A., & Elith, J. (2020). *Developing pragmatic maps of establishment likelihood for plant pests* [Technical Report for CEBRA project 170607]. University of Melbourne. http://rgdoi.net/10.13140/RG.2.2.21964.23685

Cameron, A. R. (2012). The consequences of risk-based surveillance: Developing output-based standards for surveillance to demonstrate freedom from disease. *Preventive Veterinary Medicine*, *105*(4), 280–286. https://doi.org/10.1016/j.prevetmed.2012.01.009

Cameron, K. H., Somachandra, K. P., Curry, C. N., Jenner, W. H., & Hobbs, S. L. A. (2016). Delivering Actionable Plant Health Knowledge to Smallholder Farmers Through the Plantwise Program. *Journal of Agricultural & Food Information*, *17*(4), 212–229. https://doi.org/10.1080/10496505.2016.1211530

Campbell, K., & Donlan, C. J. (2005). Feral Goat Eradications on Islands. *Conservation Biology*, *19*(5), 1362–1374. https://doi.org/10.1111/j.1523-1739.2005.00228.x

Campbell, M. L., Gould, B., & Hewitt, C. L. (2007). Survey evaluations to assess marine bioinvasions. *Marine Pollution Bulletin*, *55*(7–9), 360–378. https://doi.org/10.1016/j.marpolbul.2007.01.015

Campbell, T. A., Foster, J. A., Bodenchuk, M. J., Eisemann, J. D., Staples, L., & Lapidge, S. J. (2013). Effectiveness and target-specificity of a novel design of food dispenser to deliver a toxin to feral swine in the United States. *International Journal of Pest Management*, *59*(3), 197–204. https://doi.org/10.1080/09670874.2013.815830

Campbell, T. A., Long, D. B., & Massei, G. (2011). Efficacy of the Boar-Operated-System to deliver baits to feral swine. *Preventive Veterinary Medicine*, *98*(4), 243–249. https://doi.org/10.1016/j.prevetmed.2010.11.018

Capel, K. C. C., Creed, J., Kitahara, M. V., Chen, C. A., & Zilberberg, C. (2019). Multiple introductions and secondary dispersion of *Tubastraea* spp. In the Southwestern Atlantic. *Scientific Reports*, *9*(1), 13978. https://doi.org/10.1038/s41598-019-50442-3

Capinha, C., Essl, F., Seebens, H., Pereira, H. M., & Kühn, I. (2018). Models of alien species richness show moderate predictive accuracy and poor transferability. *NeoBiota*, *38*, 77–96. https://doi.org/10.3897/neobiota.38.23518

Capizzi, D., Baccetti, N., & Sposimo, P. (2016). Fifteen Years of Rat Eradication on Italian Islands. In F. M. Angelici (Ed.), *Problematic Wildlife: A Cross-Disciplinary Approach* (pp. 205–227). Springer International Publishing. https://doi.org/10.1007/978-3-319-22246-2_10

Caplat, P., Coutts, S., & Buckley, Y. M. (2012). Modeling population dynamics, landscape structure, and management decisions for controlling the spread of invasive plants. *Annals of the New York Academy of Sciences*, *1249*(1), 72–83. https://doi.org/10.1111/j.1749-6632.2011.06313.x

Carim, K. J., Bean, N. J., Connor, J. M., Baker, W. P., Jaeger, M., Ruggles, M. P., McKelvey, K. S., Franklin, T. W., Young, M. K., & Schwartz, M. K. (2020). Environmental DNA Sampling Informs Fish Eradication Efforts: Case Studies and Lessons Learned. *North American Journal of Fisheries Management*, *40*(2), 488–508. https://doi.org/10.1002/nafm.10428

Carlton, J. T. (1996). Marine bioinvasions: The alteration of marine ecosystems by nonindigenous species. *Oceanography*, *9*(1), 36–43. https://doi.org/10.5670/oceanog.1996.25

Carlton, J. T., Keith, I., & Ruiz, G. M. (2019). Assessing marine bioinvasions in the Galápagos Islands: Implications for conservation biology and marine protected areas. *Aquatic Invasions*, *14*(1), 1–20. https://doi.org/10.3391/ai.2019.14.1.01

Carrion, M., & Madoff, L. C. (2017). ProMED-mail: 22 years of digital surveillance of emerging infectious diseases. *International Health*, *9*(3), 177–183. https://doi.org/10.1093/inthealth/ihx014

Carrion, V., Donlan, C. J., Campbell, K. J., Lavoie, C., & Cruz, F. (2011). Archipelago-Wide Island Restoration in the Galapagos Islands: Reducing Costs of Invasive Mammal Eradication Programs and Reinvasion Risk. *Plos ONE*, *6*(5), e18835. https://doi.org/10.1371/journal.pone.0018835

Carter, A., Barr, S., Bond, C., Paske, G., Peters, D., & van Dam, R. (2016). Controlling sympatric pest mammal populations in New Zealand with self-resetting, toxicant-free traps: A promising tool for invasive species management. *Biological Invasions*, *18*(6), 1723–1736. https://doi.org/10.1007/s10530-016-1115-4

Carvalheiro, L. G., Buckley, Y. M., Ventim, R., Fowler, S. V., & Memmott, J. (2008). Apparent competition can compromise the safety of highly specific biocontrol agents. *Ecology Letters*, *11*(7), 690–700. https://doi.org/10.1111/j.1461-0248.2008.01184.x

Carwardine, J., Kottege, N., Borges, P., Nicol, S., Merz, T., Bandyopadhyay, T., & Sheppard, A. (2016). *Scoping study final report on the viability of robotics for the widespread management of prickly acacia* (p. 26). CSIRO. http://hdl.handle.net/102.100.100/90148?index=1

Carwardine, J., Martin, T. G., Firn, J., Reyes, R. P., Nicol, S., Reeson, A., Grantham, H. S., Stratford, D., Kehoe, L., & Chadès, I. (2019). Priority Threat Management for biodiversity conservation: A handbook. *Journal of Applied Ecology*, *56*(2), 481–490. https://doi.org/10.1111/1365-2664.13268

Carwardine, J., O'Connor, T., Legge, S., Mackey, B., Possingham, H. P., & Martin, T. G. (2012). Prioritizing threat management for biodiversity conservation. *Conservation Letters*, *5*(3), 196–204. https://doi.org/10.1111/j.1755-263X.2012.00228.x

Castelblanco-Martínez, D. N., Moreno-Arias, R. A., Velasco, J. A., Moreno-Bernal, J. W., Restrepo, S., Noguera-Urbano, E. A., Baptiste, M. P., García-Loaiza, L. M., & Jiménez, G. (2021). A hippo in the room: Predicting the persistence and dispersion of an invasive mega-vertebrate in Colombia, South America. *Biological Conservation*, *253*, 108923. https://doi.org/10.1016/j.biocon.2020.108923

Castillo, M. L. (2019). *Processes and drivers of* Prosopis *invasions in Eastern Africa* [Thesis]. Stellenbosch : Stellenbosch University.

Castrejón, M., Defeo, O., Reck, G., & Charles, A. (2014). Fishery Science in Galapagos: From a Resource-Focused to a Social–Ecological Systems Approach. In J. Denkinger & L. Vinueza (Eds.), *The Galapagos Marine Reserve: A Dynamic Social-Ecological System* (pp. 159–185). Springer International Publishing. https://doi.org/10.1007/978-3-319-02769-2_8

Castriciones, E. V., & Vijayan, V. (2020). Biosecurity risk mapping and gap analysis in South East Asia. *Journal of Biosafety and Biosecurity*, *2*(1), 36–43. https://doi.org/10.1016/j.jobb.2020.03.001

Castro, K. L., Giachetti, C. B., Battini, N., Bortolus, A., & Schwindt, E. (2020). Cleaning by beaching: Introducing a new alternative for hull biofouling management in Argentina. *Aquatic Invasions*, *15*(1), 63–80. https://doi.org/10.3391/ai.2020.15.1.05

Catford, J. A., Downes, B. J., Gippel, C. J., & Vesk, P. A. (2011). Flow regulation reduces native plant cover and facilitates exotic invasion in riparian wetlands. *Journal of Applied Ecology*, *48*(2), 432–442. https://doi.org/10.1111/j.1365-2664.2010.01945.x

Catford, J. A., Morris, W. K., Vesk, P. A., Gippel, C. J., & Downes, B. J. (2014). Species and environmental characteristics point to flow regulation and drought as drivers of riparian plant invasion. *Diversity and Distributions*, *20*(9), 1084–1096. https://doi.org/10.1111/ddi.12225

Caton, B. P., Koop, A. L., Fowler, L., Newton, L., & Kohl, L. (2018). Quantitative Uncertainty Analysis for a Weed Risk Assessment System. *Risk Analysis*, *38*(9), 1972–1987. https://doi.org/10.1111/risa.12979

Caughley, G., Pech, R., & Grice, D. (1992). Effect of fertility control on a population's productivity. *Wildlife Research*, *19*(6), 623–627. https://doi.org/10.1071/wr9920623

Caut, S., Angulo, E., & Courchamp, F. (2009). Avoiding surprise effects on Surprise Island: Alien species control in a multitrophic level perspective. *Biological Invasions*, *11*(7), 1689–1703. https://doi.org/10.1007/s10530-008-9397-9

Cayot, L. J., Campbell, K., & Carrión, V. (2021). Chapter 19 – Invasive species: Impacts, control, and eradication. In J. P. Gibbs, L. J. Cayot, & W. T. Aguilera (Eds.), *Galapagos Giant Tortoises* (pp. 381–399). Academic Press. https://doi.org/10.1016/B978-0-12-817554-5.00009-5

CBD. (2014). *Pathways of introduction of invasive species, their prioritization and management*. 18. https://www.cbd.int/doc/meetings/sbstta/sbstta-18/official/sbstta-18-09-add1-en.pdf

CBD. (2018). *Invasive alien species: The application of classical biological control for the management of established invasive alien species causing environmental impacts*. Fourteenth meeting of Conference of the parties to the Convention on Biological Diversity, Sharm El-Sheikh, Egypt. https://www.cbd.int/doc/c/0c6f/7a35/eb8815eff54c3bc4a02139fd/cop-14-inf-09-en.pdf

CBD. (2019). *Synthesis report of the online forum on invasive alien species*. *Ad hoc* technical expert group on invasive alien species, Montreal, Canada. https://www.cbd.int/doc/c/d56b/254f/f263e27be6e1bb97f564e21d/ias-ahteg-2019-01-inf-01-en.pdf

CBD. (2020a). *Preparations for the Post-2020 Biodiversity Framework*. Convention on Biological Diversity. https://www.cbd.int/conferences/post2020

CBD. (2020b, July 10). *Invasive alien species*. Twenty-fourth meeting of Subsidiary body on scientific technical and technological advice, Quebec City, Canada. https://www.cbd.int/doc/c/4e0e/0677/296c40f85b26a582b8116160/sbstta-24-10-en.pdf

CBD. (2022a). 15/4. Kunming-Montreal Global Biodiversity Framework. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Fifteenth meeting (Part II) of Conference of the parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf

CBD. (2022b). 15/27. Invasive alien species. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Fifteenth meeting of Conference of the parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-27-en.pdf

CBD. (2022c). *Invasive alien species: Draft decision submitted by the Chair*. Fifteenth meeting of Conference of Parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/c/ce5d/79c3/176ae7eb5c434bf4dcd42eac/cop-15-wg2-crp-05-en.pdf

Centre for Invasive Species Solutions. (2021). Tools for developing cost-effective decisions for managing invasive pest eradications. *Centre for Invasive Species Solutions*. https://invasives.com.au/research/tools-developing-cost-effective-decisions-managing-invasive-pest-eradications/

Champer, J., Kim, I. K., Champer, S. E., Clark, A. G., & Messer, P. W. (2021). Suppression gene drive in continuous space can result in unstable persistence of both drive and wild-type alleles. *Molecular Ecology*, *30*(4), 1086–1101. https://doi.org/10.1111/mec.15788

Chandrasekaran, S., & Swamy, P. S. (2016). Ecological and Socioeconomic Impacts of *Prosopis juliflora* Invasion in the Semiarid Ecosystems in Selected Villages of Ramnad District in Tamil Nadu. In N. Ghosh, P. Mukhopadhyay, A. Shah, & M. Panda (Eds.), *Nature, Economy and Society: Understanding the Linkages* (pp. 347–357). Springer, India. https://doi.org/10.1007/978-81-322-2404-4_18

Chapman, D., Pescott, O. L., Roy, H. E., & Tanner, R. (2019). Improving species distribution models for invasive non-native species with biologically informed pseudo-absence selection. *Journal of Biogeography*, *46*(5), 1029–1040. https://doi.org/10.1111/jbi.13555

Charles, J. G., Avila, G. A., Hoelmer, K. A., Hunt, S., Gardner-Gee, R., MacDonald, F., & Davis, V. (2019). Experimental assessment of the biosafety of *Trissolcus japonicus* in New Zealand, prior to the anticipated arrival of the invasive pest *Halyomorpha halys*. *BioControl*, *64*(4), 367–379. https://doi.org/10.1007/s10526-019-09949-x

Chaudhry, M. Q. (1997). Review A Review of the Mechanisms Involved in the Action of Phosphine as an Insecticide and Phosphine Resistance in Stored-Product Insects. *Pesticide Science*, *49*(3), 213–228. https://doi.org/10.1002/(SICI)1096-9063(199703)49:3%3C213::AID-PS516%3E3.0.CO;2-%23

Chen, J.-W., Lin, W.-J., Cheng, H.-J., Hung, C.-L., Lin, C.-Y., & Chen, S.-P. (2021). A Smartphone-Based Application for Scale Pest Detection Using Multiple-Object Detection Methods. *Electronics*, *10*(4), 372. https://doi.org/10.3390/electronics10040372

Chikoye, D., Ellis-Jones, J., Tarawali, G., Kormawa, P., Nielsen, O., Ibana, S., & Avav, T.-R. (2006). Farmers' perceptions of the speargrass (*Imperata cylindrica*) problem and its control in the lowland sub-humid savannah of Nigeria. *Journal of Food, Agriculture & Environment*, *4*(3 & 4), 118–126. https://www.academia.edu/48826875/Farmers_perceptions_of_the_speargrass_Imperata_cylindrica_problem_and_its_control_in_the_lowland_sub_humid_savannah_of_Nigeria

Chimweta, M., Nyakudya, I. W., Jimu, L., & Mashingaidze, A. B. (2020). Fall armyworm [*Spodoptera frugiperda* (J.E. Smith)] damage in maize: Management options for flood-recession cropping smallholder farmers. *International Journal of Pest Management*, *66*(2), 142–154. https://doi.org/10.1080/09670874.2019.1577514

Chizinski, C. J., Pope, K. L., & Wilde, G. R. (2010). A modelling approach to evaluate potential management actions designed to increase growth of white perch in a high-density population. *Fisheries Management and Ecology*, *17*(3), 262–271. https://doi.org/10.1111/j.1365-2400.2009.00723.x

Choo, A., Crisp, P., Saint, R., O'Keefe, L. V., & Baxter, S. W. (2018). CRISPR/Cas9-mediated mutagenesis of the *white* gene in the tephritid pest *Bactrocera tryoni*. *Journal of Applied Entomology*, *142*(1–2), 52–58. https://doi.org/10.1111/jen.12411

Chowdhury, M. E. H., Rahman, T., Khandakar, A., Mazhar, R., Kadir, M. A., Mahbub, Z. B., Islam, K. R., Khan, M. S., Iqbal, A., Emadi, N. A., Reaz, M. B. I., & Islam, M. T. (2020). Can AI Help in Screening Viral and COVID-19 Pneumonia? *IEEE Access*, *8*, 132665–132676. https://doi.org/10.1109/ACCESS.2020.3010287

Chown, S. L., Huiskes, A. H. L., Gremmen, N. J. M., Lee, J. E., Terauds, A., Crosbie, K., Frenot, Y., Hughes, K. A., Imura, S., Kiefer, K., Lebouvier, M., Raymond, B., Tsujimoto, M., Ware, C., Van de Vijver, B., & Bergstrom, D. M. (2012). Continent-wide risk assessment for the establishment of nonindigenous species in Antarctica. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(13), 4938–4943. https://doi.org/10.1073/pnas.1119787109

Chua, S. D. X., Lu, X. X., Oeurng, C., Sok, T., & Grundy-Warr, C. (2022). Drastic decline of flood pulse in the Cambodian floodplains (Mekong River and Tonle Sap system). *Hydrology and Earth System Sciences*, *26*(3), 609–625. https://doi.org/10.5194/hess-26-609-2022

CIRCABC. (2021). *Scientific Forum on Alien Invasive Species*. Communication and Information Resource Centre for Administrations, Businesses and Citizens. https://circabc.europa.eu/ui/group/98665af0-7dfa-448c-8bf4-e1e086b50d2c/library/ed95cea1-4f6a-4a3b-b27d-b2bfb8288c42

Clarke, D. A., Palmer, D. J., McGrannachan, C., Burgess, T. I., Chown, S. L., Clarke, R. H., Kumschick, S., Lach, L., Liebhold, A. M., Roy, H. E., Saunders, M. E., Yeates, D. K., Zalucki, M. P., & McGeoch, M. A. (2021). Options for reducing uncertainty in impact classification for alien species. *Ecosphere*, *12*(4), e03461. https://doi.org/10.1002/ecs2.3461

Clarke, S., Stenekes, N., Kancans, R., Woodland, C., & Robinson, A. (2018). Undelivered risk: A counter-factual analysis of the biosecurity risk avoided by inspecting international mail articles. *NeoBiota*, *40*, 73–86. https://doi.org/10.3897/neobiota.40.28840

Clement, R., Dunbabin, M., & Wyeth, G. (2005). Toward robust image detection of crown-of-thorns starfish for autonomous population monitoring. In C. Sammut (Ed.), *Proceedings of the 2005 Australasian Conference on Robotics and Automation* (pp. 1–8). Australian Robotics and Automation Association Inc. http://www.araa.asn.au/acra/acra2005/papers/clement.pdf

Clements, E. J., & Foster, M. C. (1994). *Alien plants of the British Isles. A provisional catalogue of vascular plants (excluding grasses)* (p. 589). Botanical Society of the British Isles. https://www.cabi.org/isc/abstract/19952313216

Cock, M. J. W., Murphy, S. T., Kairo, M. T. K., Thompson, E., Murphy, R. J., & Francis, A. W. (2016). Trends in the classical biological control of insect pests by insects: An update of the BIOCAT database. *BioControl*, *61*(4), 349–363. https://doi.org/10.1007/s10526-016-9726-3

Collier, N., Doan, S., Kawazoe, A., Goodwin, R. M., Conway, M., Tateno, Y., Ngo, Q.-H., Dien, D., Kawtrakul, A., Takeuchi, K., Shigematsu, M., & Taniguchi, K. (2008). BioCaster: Detecting public health rumors with a Web-based text mining system. *Bioinformatics*, *24*(24), 2940–2941. https://doi.org/10.1093/bioinformatics/btn534

Collins, P. J., Emery, R. N., & Wallbank, B. E. (2003). Two decades of monitoring and managing phosphine resistance in Australia. In P. F. Credland, D. M. Armitage, C. H. Bell, P. M. Cogan, & E. Highley (Eds.), *Proceedings of the 8th International Working Conference on Stored Product Protection, York, UK, 22-26 July 2002* (pp. 570–575). CABI. https://doi.org/10.1079/9780851996912.0570

Colunga-Garcia, M., Haack, R. A., Magarey, R. D., & Borchert, D. M. (2013). Understanding trade pathways to target biosecurity surveillance. *NeoBiota*, *18*, 103–118. https://doi.org/10.3897/neobiota.18.4019

Conser, C., Seebacher, L., Fujino, D. W., Reichard, S., & DiTomaso, J. M. (2015). The Development of a Plant Risk Evaluation (PRE) Tool for Assessing the Invasive Potential of Ornamental Plants. *PLoS ONE*, *10*(3), e0121053. https://doi.org/10.1371/journal.pone.0121053

Convey, P., & Lebouvier, M. (2009). Environmental change and human impacts on terrestrial ecosystems of the sub-Antarctic islands between their discovery and the mid-twentieth century. *Papers and Proceedings of the Royal Society of Tasmania*, *143*(1), 33–44. https://doi.org/10.26749/rstpp.143.1.33

Cook, D., & Proctor, W. (2007). Assessing the threat of exotic plant pests. *Ecological Economics*, *63*(2–3), 594–604. https://doi.org/10.1016/j.ecolecon.2006.12.021

Cook, G. D., & Dias, L. (2006). It was no accident: Deliberate plant introductions by Australian government agencies during the 20th century. *Australian Journal of Botany*, *54*(7), 601–625. https://doi.org/10.1071/BT05157

Cooke, B. D., Chudleigh, P., Simpson, S., & Saunders, G. (2013). The Economic Benefits of the Biological Control of Rabbits in Australia, 1950–2011. *Australian Economic History Review*, *53*(1), 91–107. https://doi.org/10.1111/aehr.12000

Cooke, B. D., & Fenner, F. (2002). Rabbit haemorrhagic disease and the biological control of wild rabbits, *Oryctolagus cuniculus*, in Australia and New Zealand. *Wildlife Research*, *29*(6), 689–706. https://doi.org/10.1071/wr02010

Copp, G. H., Garthwaite, R., & Gozlan, R. E. (2005). Risk identification and assessment of non-native freshwater fishes: A summary of concepts and perspectives on protocols for the UK. *Journal of Applied Ichthyology*, *21*(4), 371–373. https://doi.org/10.1111/j.1439-0426.2005.00692.x

Copp, G. H., Russell, I. C., Peeler, E. J., Gherardi, F., Tricarico, E., Macleod, A., Cowx, I. G., Nunn, A. D., Occhipinti-Ambrogi, A., Savini, D., Mumford, J., & Britton, J. R. (2016). European Non-native Species in Aquaculture Risk Analysis Scheme—A summary of assessment protocols and decision support tools for use of alien species in aquaculture. *Fisheries Management and Ecology*, *23*(1), 1–11. https://doi.org/10.1111/fme.12074

Copp, G. H., Templeton, M., & Gozlan, R. E. (2007). Propagule pressure and the invasion risks of non-native freshwater fishes: A case study in England. *Journal of Fish Biology*, *71*(sd), 148–159. https://doi.org/10.1111/j.1095-8649.2007.01680.x

Corn, M. L., & Johnson, R. (2013). *Invasive Species: Major Laws and the Role of Selected Federal Agencies* (p. 58). Congressional Research Service. https://digital.library.unt.edu/ark:/67531/metadc813137/m1/1/

Costello, C., Springborn, M., McAusland, C., & Solow, A. (2007). Unintended biological invasions: Does risk vary by trading partner? *Journal of Environmental Economics and Management*, *54*(3), 262–276. https://doi.org/10.1016/j.jeem.2007.06.001

Council of Europe. (2021). *Group of experts on Invasive Alien Species (IAS)*. Convention on the Conservation of European Wildlife and Natural Habitats. https://www.coe.int/en/web/bern-convention/on-invasive-alien-species

Courchamp, F., Fournier, A., Bellard, C., Bertelsmeier, C., Bonnaud, E., Jeschke, J. M., & Russell, J. C. (2017). Invasion Biology: Specific Problems and Possible Solutions. *Trends in Ecology & Evolution*, *32*(1), 13–22. https://doi.org/10.1016/j.tree.2016.11.001

Courtois, P., Figuieres, C., Mulier, C., & Weill, J. (2018). A Cost–Benefit Approach for Prioritizing Invasive Species. *Ecological Economics*, *146*, 607–620. https://doi.org/10.1016/j.ecolecon.2017.11.037

Cowan, D. P., van der Waal, Z., Pidcock, S., Gomm, M., Stephens, N., Brash, M., White, P. C. L., Mair, L., & Mill, A. C. (2020). Adaptive management of an iconic invasive goat *Capra hircus* population. *Mammal Review*, *50*(2), 180–186. https://doi.org/10.1111/mam.12176

Cowled, B. D., Sergeant, E. S. G., Leslie, E. E. C., Crosbie, A., Burroughs, A., Kingston, O., Neill, M., Sawford, K., & van Andel, M. (2022). Use of scenario tree modelling to plan freedom from infection surveillance: *Mycoplasma bovis* in New Zealand. *Preventive Veterinary Medicine*, *198*, 105523. https://doi.org/10.1016/j.prevetmed.2021.105523

Cox, T., Strive, T., Mutze, G., West, P., & Saunders, G. (2013). *Benefits of Rabbit biocontrol in Australia*. Invasive Animals Cooperative Research Centre. https://www.pestsmart.org.au/wp-content/uploads/2014/03/RabbitBiocontrol.pdf

Crall, A. W., Newman, G. J., Stohlgren, T. J., Holfelder, K. A., Graham, J., & Waller, D. M. (2011). Assessing citizen science data quality: An invasive species case study. *Conservation Letters*, *4*(6), 433–442. https://doi.org/10.1111/j.1755-263X.2011.00196.x

Cranmer, C., Scott, A., Thongsavath, O., & Xeuasing, K. (2018). *Final report: Climate Change Vulnerability Assessment, Beung Kiat Ngong Ramsar Site, Lao PDR.* (p. 77). International Union for Conservation of Nature. https://www.iucn.org/sites/default/files/content/documents/2019/climate_change_vulnerability_assessment_beung_kiat_ngong_ramsar_site_lao_pdr.pdf

Creed, J. C., Casares, F. A., Oigman-Pszczol, S. S., & Masi, B. P. (2021). Multi-site experiments demonstrate that control of invasive corals (*Tubastraea* spp.) by manual removal is effective. *Ocean & Coastal Management*, *207*, 105616. https://doi.org/10.1016/j.ocecoaman.2021.105616

Creed, J. C., Junqueira, A., Fleury, B. G., Mantelatto, M. C., & Oigman-Pszczol, S. S. (2017). The Sun-Coral Project: The first social-environmental initiative to manage the biological invasion of *Tubastraea* spp. in Brazil. *Management of Biological Invasions*, *8*(2), 181–195. https://doi.org/10.3391/mbi.2017.8.2.06

Crowley, S. L., Hinchliffe, S., & McDonald, R. A. (2019). The parakeet protectors: Understanding opposition to introduced species management. *Journal of Environmental Management*, *229*, 120–132. https://doi.org/10.1016/j.jenvman.2017.11.036

Cruz, F., Carrion, V., Campbell, K. J., Lavoie, C., & Donlan, C. J. (2009). Bio-Economics of Large-Scale Eradication of Feral Goats From Santiago Island, Galápagos. *Journal of Wildlife Management*, *73*(2), 191–200. https://doi.org/10.2193/2007-551

CSIRO. (2020). *Australia's Biosecurity Future. Unlocking the next decade of resilience (2020–2030)* (p. 48). Commonwealth Scientific and Industrial Research Organisation. https://www.csiro.au/-/media/Do-Business/Files/Futures/Biosecurity/20-00277_SER-FUT_REPORT_BiosecurityFutures_WEB_201028.pdf

Csiszár, Á., & Korda, M. (Eds.). (2015). *Practical experiences in invasive alien plant control* (Vol. 3). Budapest: Duna–Ipoly National Park Directorate. https://www.dunaipoly.hu/uploads/2016-02/20160202200313-rosalia-handbook-ver2-6xtoafsq.pdf

Cuan, K., Zhang, T., Huang, J., Fang, C., & Guan, Y. (2020). Detection of avian influenza-infected chickens based on a chicken sound convolutional neural network. *Computers and Electronics in Agriculture*, *178*, 105688. https://doi.org/10.1016/j.compag.2020.105688

Cui, S., Ling, P., Zhu, H., & Keener, H. M. (2018). Plant Pest Detection Using an Artificial Nose System: A Review. *Sensors*, *18*(2), 378. https://doi.org/10.3390/s18020378

Dafforn, K. (2017). Eco-engineering and management strategies for marine infrastructure to reduce establishment and dispersal of non-indigenous species. *Management of Biological Invasions*, *8*(2), 153–161. https://doi.org/10.3391/mbi.2017.8.2.03

Daigneault, A. J., & Brown, P. (2013). Invasive species management in the Pacific using survey data and benefit-cost analysis. *57th Australian Agricultural & Resource Economics Society Annual Conference*. https://doi.org/10.22004/AG.ECON.152140

Dana, E. D., García-de-Lomas, J., Verloove, F., & Vilà, M. (2019). Common deficiencies of actions for managing invasive alien species: A decision-support checklist. *NeoBiota*, *48*, 97–112. https://doi.org/10.3897/neobiota.48.35118

D'Antonio, C. M., August-Schmidt, E., & Fernandez-Going, B. (2016). Invasive Species and Restoration Challenges. In M. A. Palmer, J. B. Zedler, & D. A. Falk (Eds.), *Foundations of Restoration Ecology* (pp. 216–244). Island Press/Center for Resource Economics. https://doi.org/10.5822/978-1-61091-698-1_8

D'Antonio, C. M., & Meyerson, L. A. (2002). Exotic plant species as problems and solutions in ecological restoration: A synthesis. *Restoration Ecology*, *10*(4), 703–713. https://doi.org/10.1046/j.1526-100X.2002.01051.x

Das, P. R., & Sherif, S. M. (2020). Application of Exogenous dsRNAs-induced RNAi in Agriculture: Challenges and Triumphs. *Frontiers in Plant Science*, *11*, 946. https://doi.org/10.3389/fpls.2020.00946

Dash, J. P., Watt, M. S., Paul, T. S. H., Morgenroth, J., & Pearse, G. D. (2019). Early Detection of Invasive Exotic Trees Using UAV and Manned Aircraft Multispectral and LiDAR Data. *Remote Sensing*, *11*(15), 1812. https://doi.org/10.3390/rs11151812

Dassonville, N., Vanderhoeven, S., Vanparys, V., Hayez, M., Gruber, W., & Meerts, P. (2008). Impacts of alien invasive plants on soil nutrients are correlated with initial site conditions in NW Europe. *Oecologia*, *157*(1), 131–140. https://doi.org/10.1007/s00442-008-1054-6

Datta, A., Schweiger, O., & Kühn, I. (2020). Origin of climatic data can determine the transferability of species distribution models. *NeoBiota*, *59*, 61–76. https://doi.org/10.3897/neobiota.59.36299

Davies, S. J., Jordaan, M. S., Karsten, M., Terblanche, J. S., Turner, A. A., van Wilgen, N. J., Veldtman, R., Zengeya, T. A., & Measey, J. (2020). Experience and Lessons from Alien and Invasive Animal Control Projects in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (pp. 629–663). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_22

Dawei, W., Limiao, D., Jiangong, N., Jiyue, G., Hongfei, Z., & Zhongzhi, H. (2019). Recognition pest by image-based transfer learning. *Journal of the Science of Food and Agriculture*, *99*(10), 4524–4531. https://doi.org/10.1002/jsfa.9689

Dawson, W., Peyton, J. M., Pescott, O. L., Adriaens, T., Cottier-Cook, E. J., Frohlich, D. S., Key, G., Malumphy, C., Martinou, A. F., Minchin, D., Moore, N., Rabitsch, W., Rorke, S. L., Tricarico, E., Turvey, K. M. A., Winfield, I. J., Barnes, D. K. A., Baum, D., Bensusan, K., … Roy, H. E. (2022). Horizon scanning for potential invasive non-native species across the United Kingdom Overseas Territories. *Conservation Letters*, *16*(1), e12928. https://doi.org/10.1111/conl.12928

Day, M. D., Clements, D. R., Gile, C., Senaratne, W. K. A. D., Shen, S., Weston, L. A., & Zhang, F. (2016). Biology and Impacts of Pacific Islands Invasive Species. 13. *Mikania micrantha* Kunth (Asteraceae). *Pacific Science*, *70*(3), 257–285. https://doi.org/10.2984/70.3.1

Day, M. D., Kawi, A. P., Fidelis, J., Tunabuna, A., Orapa, W., Swamy, B., Ratutini, J., Saul-Maora, J., & Dewhurst, C. F. (2011). Biology, Field Release and Monitoring of the Rust Fungus *Puccinia spegazzinii* (Pucciniales: Pucciniaceae), a Biological Control Agent of *Mikania micrantha* (Asteraceae) in Papua New Guinea and Fiji. *Proceedings of the XIII International Symposium on Biological Control of Weeds*, 211–217. http://bugwoodcloud.org/ibiocontrol/proceedings/pdf/Day.pdf

Day, M. D., Kawi, A., Tunabuna, A., Fidelis, J., Swamy, B., Ratutuni, J., Saul-Maora, J., Dewhurst, C. F., & Orapa, W. (2011). The distribution and socio-economic impacts of *Mikania micrantha* (Asteraceae) in Papua New Guinea and Fiji and prospects for its biocontrol. *Proceedings of the 23rd Asian-Pacific Weed Science Society Conference. Asian-Pacific Weed Science Society*, *1*, 146–153. https://researchoutput.csu.edu.au/files/9704340/29497_Weston_23rd%20APWSS%20Conference%20Proceedings%20Vol%201.pdf

Day, M. D., & Witt, A. B. (2019). Weed biological control: Challenges and opportunities. *Weeds-Journal of the Asian-Pacific Weed Science Society*, *1*(2), 34–44. https://library.sprep.org/sites/default/files/2021-05/weed-biological-challenges-opportunities.pdf

Dayoub, F., Dunbabin, M., & Corke, P. (2015). Robotic detection and tracking of Crown-of-Thorns starfish. *2015 IEEE/RSJ International Conference on Intelligent Robots and Systems (IROS)*, 1921–1928. https://doi.org/10.1109/IROS.2015.7353629

De Fine Licht, J. (2014). Transparency actually: How transparency affects public perceptions of political decision-making. *European Political Science Review*, *6*(2), 309–330. https://doi.org/10.1017/S1755773913000131

de Oliveira Soares, M., Davis, M., & de Macêdo Carneiro, P. B. (2018). Northward range expansion of the invasive coral (*Tubastraea tagusensis*) in the southwestern Atlantic. *Marine Biodiversity*, *48*(3), 1651–1654. https://doi.org/10.1007/s12526-016-0623-x

De Poorter, M. (2007). *Invasive alien species and protected areas: A scoping report part 2*. The Global Invasive Species Programme.

Deak, B. P., Ostendorf, B., Taggart, D. A., Peacock, D. E., & Bardsley, D. K. (2019). The Significance of Social Perceptions in Implementing Successful Feral Cat Management Strategies: A Global Review. *Animals*, *9*(9), 617. https://doi.org/10.3390/ani9090617

Dechoum, M. de S., Giehl, E. L. H., Sühs, R. B., Silveira, T. C. L., & Ziller, S. R. (2019). Citizen engagement in the management of non-native invasive pines: Does it make a difference? *Biological Invasions*, *21*(1), 175–188. https://doi.org/10.1007/s10530-018-1814-0

DEFRA. (2019). *Comprehensive analysis of pathways of unintentional introduction and spread of invasive alien species – report of the UK*. Department for Environment Food and Rural Affairs, London, UK. https://www.semanticscholar.org/paper/Comprehensive-analysis-of-pathways-of-unintentional/279756815617296a2f2a3161953525352b6aba40

del Rocio Amezcua, M., Pearl, D. L., Friendship, R. M., & McNab, W. B. (2010). Evaluation of a veterinary-based syndromic surveillance system implemented for swine. *Canadian Journal of Veterinary Research = Revue Canadienne De Recherche Veterinaire*, *74*(4), 241–251. https://pubmed.ncbi.nlm.nih.gov/21197223/

Delane, R. (2019). *Independent Assurance Review of New Zealand's biosecurity international border defences for passenger and mail pathways* (pp. 1–36). Ministry for Primary Industries. https://apo.org.au/sites/default/files/resource-files/2019-04/apo-nid233256.pdf

Delaney, D. G., Sperling, C. D., Adams, C. S., & Leung, B. (2008). Marine invasive species: Validation of citizen science and implications for national monitoring networks. *Biological Invasions*, *10*(1), 117–128. https://doi.org/10.1007/s10530-007-9114-0

Demertzis, K., Iliadis, L., & Anezakis, V.-D. (2017). A deep spiking machine-hearing system for the case of invasive fish species. *2017 IEEE International Conference on Innovations in Intelligent Systems and Applications (INISTA)*, 23–28. https://doi.org/10.1109/INISTA.2017.8001126

Demertzis, K., Iliadis, L. S., & Anezakis, V.-D. (2018). Extreme deep learning in biosecurity: The case of machine hearing for marine species identification. *Journal of Information and Telecommunication*, *2*(4), 492–510. https://doi.org/10.1080/24751839.2018.1501542

Department of Primary Industries, New South Wales. (2017). *NSW Weed Risk Management system—Background information*. https://www.dpi.nsw.gov.au/biosecurity/weeds/strategy/nsw-weed-risk-management-system/background-information

Descamps, S., & De Vocht, A. (2017). The sterile male release approach as a method to control invasive amphibian populations: A preliminary study on *Lithobates catesbeianus*. *Management of Biological Invasions*, *8*(3), 361–370. https://doi.org/10.3391/mbi.2017.8.3.09

Devitt, S. K., Perez, T., Polson, D., Pearce, T. R., Quagliata, R., Taylor, W., Thornby, D., & Beekhuyzen, J. (2017). A cognitive decision tool to optimise integrated weed management. *Proceedings of the 7th Asian-Australasian Conference on Precision Agriculture*, 1–8. https://doi.org/10.5281/zenodo.895617

Devorshak, C. (2012). *Plant Pest Risk Analysis: Concepts and Applications*. CABI.

DFO. (2021, December 10). *Science Advisory Report 2021/043*. Fisheries and Oceans Canada, Government of Canada. https://www.dfo-mpo.gc.ca/csas-sccs/Publications/SAR-AS/2021/2021_043-eng.html

D'hondt, B., Vanderhoeven, S., Roelandt, S., Mayer, F., Versteirt, V., Adriaens, T.,

Ducheyne, E., San Martin, G., Grégoire, J.-C., Stiers, I., Quoilin, S., Cigar, J., Heughebaert, A., & Branquart, E. (2015). *Harmonia+* and *Pandora+*: Risk screening tools for potentially invasive plants, animals and their pathogens. *Biological Invasions*, *17*(6), 1869–1883. https://doi.org/10.1007/s10530-015-0843-1

Di Marco, M., Baker, M. L., Daszak, P., De Barro, P., Eskew, E. A., Godde, C. M., Harwood, T. D., Herrero, M., Hoskins, A. J., Johnson, E., Karesh, W. B., Machalaba, C., Garcia, J. N., Paini, D., Pirzl, R., Smith, M. S., Zambrana-Torrelio, C., & Ferrier, S. (2020). Opinion: Sustainable development must account for pandemic risk. *Proceedings of the National Academy of Sciences*, *117*(8), 3888–3892. https://doi.org/10.1073/pnas.2001655117

Diagne, C., Leroy, B., Gozlan, R. E., Vaissière, A.-C., Assailly, C., Nuninger, L., Roiz, D., Jourdain, F., Jarić, I., & Courchamp, F. (2020). InvaCost, a public database of the economic costs of biological invasions worldwide. *Scientific Data*, *7*(1), 277. https://doi.org/10.1038/s41597-020-00586-z

Diagne, C., Leroy, B., Vaissière, A.-C., Gozlan, R. E., Roiz, D., Jarić, I., Salles, J.-M., Bradshaw, C. J. A., & Courchamp, F. (2021). High and rising economic costs of biological invasions worldwide. *Nature*, *592*(7855), 571–576. https://doi.org/10.1038/s41586-021-03405-6

Dias, P. J., Lukehurst, S. S., Simpson, T., Rocha, R. M., Tovar-Hernández, M. A., Wellington, C., McDonald, J. I., Snow, M., & Kennington, W. J. (2021). Multiple introductions and regional spread shape the distribution of the cryptic ascidian *Didemnum perlucidum* in Australia: An important baseline for management under climate change. *Aquatic Invasions*, *16*(2), 297–313. https://doi.org/10.3391/ai.2021.16.2.06

Dicks, L. V., Walsh, J. C., & Sutherland, W. J. (2014). Organising evidence for environmental management decisions: A '4S' hierarchy. *Trends in Ecology & Evolution*, *29*(11), 607–613. https://doi.org/10.1016/j.tree.2014.09.004

Diez, J. M., D'Antonio, C. M., Dukes, J. S., Grosholz, E. D., Olden, J. D., Sorte, C. J., Blumenthal, D. M., Bradley, B. A., Early, R., Ibáñez, I., Jones, S. J., Lawler, J. J., & Miller, L. P. (2012). Will extreme climatic events facilitate biological invasions? *Frontiers in Ecology and the Environment*, *10*(5), 249–257. https://doi.org/10.1890/110137

Dinis, M., Vicente, J. R., César de Sá, N., López-Núñez, F. A., Marchante, E., &

Marchante, H. (2020). Can Niche Dynamics and Distribution Modeling Predict the Success of Invasive Species Management Using Biocontrol? Insights from *Acacia longifolia* in Portugal. *Frontiers in Ecology and Evolution*, *8*. https://doi.org/10.3389/fevo.2020.576667

DiTomaso, J. M. (2000). Invasive weeds in rangelands: Species, impacts, and management. *Weed Science*, *48*(2), 255–265. https://doi.org/10.1614/0043-1745(2000)048[0255:IWIRSI]2.0.CO;2

Dixon, L. K., Stahl, K., Jori, F., Vial, L., & Pfeiffer, D. U. (2020). African Swine Fever Epidemiology and Control. *Annual Review of Animal Biosciences*, *8*(1), 221–246. https://doi.org/10.1146/annurev-animal-021419-083741

Dobson, A. P., Pimm, S. L., Hannah, L., Kaufman, L., Ahumada, J. A., Ando, A. W., Bernstein, A., Busch, J., Daszak, P., Engelmann, J., Kinnaird, M. F., Li, B. V., Loch-Temzelides, T., Lovejoy, T., Nowak, K., Roehrdanz, P. R., & Vale, M. M. (2020). Ecology and economics for pandemic prevention. *Science*, *369*(6502), 379–381. https://doi.org/10.1126/science.abc3189

Dodd, A., Stoeckl, N., Baumgartner, J., & Kompas, T. (2020). *Key Result Summary: Valuing Australia's Biosecurity System* (CEBRA Project 170713; p. 41). Centre of Excellence for Biosecurity Risk Analysis. https://cebra.unimelb.edu.au/__data/assets/pdf_file/0020/3535013/CEBRA_Value_Docs_KeyResultSummary_v0.6_Endorsed.pdf

Doherty, T. S., Dickman, C. R., Johnson, C. N., Legge, S. M., Ritchie, E. G., & Woinarski, J. C. Z. (2017). Impacts and management of feral cats *Felis catus* in Australia. *Mammal Review*, *47*(2), 83–97. https://doi.org/10.1111/mam.12080

Dolezel, M., Lüthi, C., & Gaugitsch, H. (2020). Beyond limits – the pitfalls of global gene drives for environmental risk assessment in the European Union. *BioRisk*, *15*, 1–29. https://doi.org/10.3897/biorisk.15.49297

Dolezel, M., Simon, S., Otto, M., Engelhard, M., & Züghart, W. (2020). *Gene Drive Organisms: Implications for the Environment and Nature Conservation* (REP-0705; A Joint Technical Report of the EPA/ENCA Interest Group on Risk Assessment and Monitoring of GMOs). Swiss Federal Office for the Environment (FOEN), Soil and Biotechnology Division, Biotechnology Section. http://www.umweltbundesamt.at

Dominiak, B., Gott, K., McIver, D., Grant, T., Gillespie, P., Worsley, P., Clift, A., & Sergeant, E. (2011). Scenario tree risk analysis of zero detections and the eradication of yellow crazy ant ('*Anoplolepis gracilipes*' (Smith)), in New South Wales, Australia. *Plant Protection Quarterly*, *26*(4), 124–129. https://search.informit.org/doi/10.3316/informit.593474062968814

Donlan, C. J., & Wilcox, C. (2007). Complexities of costing eradications. *Animal Conservation*, *10*(2), 154–156. https://doi.org/10.1111/j.1469-1795.2007.00101.x

Downey, P. O. (2010). Managing Widespread, Alien Plant Species to Ensure Biodiversity Conservation: A Case Study Using an 11-Step Planning Process. *Invasive Plant Science and Management*, *3*(4), 451–461. https://doi.org/10.1614/IPSM-D-10-00012.1

Downey, P. O. (2013). Protecting Biodiversity Through Strategic Alien Plant Management: An Approach for Increasing Conservation Outcomes in Protected Areas. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas* (Vol. 7, pp. 507–528). Springer. https://link.springer.com/chapter/10.1007/978-94-007-7750-7_23

Downey, P. O., & Paterson, I. D. (2016). Encompassing the relative non-target risks from agents and their alien plant targets in biological control assessments. *BioControl*, *61*(6), 615–630. https://doi.org/10.1007/s10526-016-9744-1

Downey, P. O., & Sheppard, A. W. (2006). Site- versus species-based approaches to weed management in Australia. In C. Preston, J. Watts H., & N. D. Crossman (Eds.), *The Proceedings of the 15th Australian Weeds Conference: Managing Weeds in a Changing Climate* (pp. 264–267). Weed Management Society of South Australia Inc. https://www.researchgate.net/publication/276411938_Site-_versus_species-based_approaches_to_weed_management_in_Australia

Downey, P. O., Williams, M. C., Whiffen, L. K., Auld, B. A., Hamilton, M. A., Burley, A. L., & Turner, P. J. (2010). Managing Alien Plants for Biodiversity Outcomes—The Need for Triage. *Invasive Plant Science and Management*, *3*(1), 1–11. https://doi.org/10.1614/IPSM-09-042.1

Drechsler, M., Touza, J., White, P. C. L., & Jones, G. (2016). Agricultural landscape structure and invasive species: The cost-effective level of crop field clustering. *Food Security*, *8*(1), 111–121. https://doi.org/10.1007/s12571-015-0539-5

Drescher, M., Epstein, G. B., Warriner, G. K., & Rooney, R. C. (2019). An investigation of the effects of conservation incentive programs on management of invasive species by private landowners. *Conservation Science and Practice*, *1*(7), e56. https://doi.org/10.1111/csp2.56

Driscoll, D. A., Catford, J. A., Barney, J. N., Hulme, P. E., Inderjit, Martin, T. G., Pauchard, A., Pyšek, P., Richardson, D. M., Riley, S., & Visser, V. (2014). New pasture plants intensify invasive species risk. *Proceedings of the National Academy of Sciences*, *111*(46), 16622–16627. https://doi.org/10.1073/pnas.1409347111

Driscoll, D. A., Worboys, G. L., Allan, H., Banks, S. C., Beeton, N. J., Cherubin, R. C., Doherty, T. S., Finlayson, C. M., Green, K., Hartley, R., Hope, G., Johnson, C. N., Lintermans, M., Mackey, B., Paull, D. J., Pittock, J., Porfirio, L. L., Ritchie, E. G., Sato, C. F., … Williams, R. M. (2019). Impacts of feral horses in the Australian Alps and evidence-based solutions. *Ecological Management & Restoration*, *20*(1), 63–72. https://doi.org/10.1111/emr.12357

Du Preez, M., Dicken, M., & Hosking, S. G. (2012). The value of tiger shark diving within the aliwal shoal marine protected area: A travel cost analysis. *South African Journal of Economics*, *80*(3), 387–399. https://doi.org/10.1111/j.1813-6982.2011.01292.x

Duda, N., Nowak, T., Hartmann, M., Schadhauser, M., Cassens, B., Wägemann, P., Nabeel, M., Ripperger, S., Herbst, S., Meyer-Wegener, K., Mayer, F., Dressler, F., Schröder-Preikschat, W., Kapitza, R., Robert, J., Thielecke, J., Weigel, R., & Kölpin, A. (2018). BATS: Adaptive Ultra Low Power Sensor Network for Animal Tracking. *Sensors*, *18*(10), 3343. https://doi.org/10.3390/s18103343

Dunn, D. W., & Follett, P. A. (2017). The sterile insect technique (SIT)-an introduction. *Entomologia Experimentalis et Applicata*, *164*(3), 151–154. https://doi.org/10.1111/eea.12619

Durr, P. A., Graham, K., & van Klinken, R. D. (2017). Sellers' Revisited: A Big Data Reassessment of Historical Outbreaks of Bluetongue and African Horse Sickness due to the Long-Distance Wind Dispersion of *Culicoides* Midges. *Frontiers in Veterinary Science*, *4*. https://doi.org/10.3389/fvets.2017.00098

Dyck, V. A., Hendrichs, J., & Robinson, A. S. (Eds.). (2005). *Sterile insect technique: Principles and practice in area-wide integrated pest management* (1st ed.). Springer. https://doi.org/10.1007/1-4020-4051-2

Dymond, J. R., Ausseil, A.-G. E., & Overton, J. McC. (2008). A landscape approach for estimating the conservation value of sites and site-based projects, with examples from New Zealand. *Ecological Economics*, *66*(2–3), 275–281. https://doi.org/10.1016/j.ecolecon.2008.03.008

Eason, C. T., Shapiro, L., Ogilvie, S., King, C., & Clout, M. (2017). Trends in the development of mammalian pest control technology in New Zealand. *New Zealand Journal of Zoology*, *44*(4), 267–304. https://doi.org/10.1080/03014223.2017.1337645

Edwards, G., Digby, D., O'Leary, P., Rafferty, D., Jensen, M., Woolnough, A., Secomb, N., Williams, M., Schwartzkopff, K., & Bryan, R. (2016). Planning and conducting aerial culling operations for feral camels. *The Rangeland Journal*, *38*(2), 153–162. https://doi.org/10.1071/RJ15100

EFSA Panel on Plant Health (PLH). (2010). Guidance on a harmonised framework for pest risk assessment and the identification and evaluation of pest risk management options by EFSA. *EFSA Journal*, *8*(2), 66. https://doi.org/10.2903/j.efsa.2010.1495

EFSA Scientific Committee & Scientific Opinion on Risk Assessment Terminology. (2012). Scientific opinion on risk assessment terminology. *EFSA Journal*, *10*(5), 2664. https://doi.org/10.2903/j.efsa.2012.2664

Elbers, A. R. W., Backx, A., Meroc, E., Gerbier, G., Staubach, C., Hendrickx, G., van der Spek, A., & Mintiens, K. (2008). Field observations during the bluetongue serotype 8 epidemic in 2006: I. Detection of first outbreaks and clinical signs in sheep and cattle in Belgium, France and the Netherlands. *Preventive Veterinary Medicine*, *87*(1–2), 21–30. https://doi.org/10.1016/j.prevetmed.2008.06.004

Elith, J. (2017). Predicting distributions of invasive species. In A. P. Robinson, T. Walshe, M. A. Burgman, & M. Nunn (Eds.), *Invasice Species Risk assessment and management* (pp. 93–129). Cambridge University Press. https://doi.org/10.1017/9781139019606.006

Elkind, K., Sankey, T. T., Munson, S. M., & Aslan, C. E. (2019). Invasive buffelgrass detection using high-resolution satellite and UAV imagery on Google Earth Engine. *Remote Sensing in Ecology and Conservation*, *5*(4), 318–331. https://doi.org/10.1002/rse2.116

Ellison, C. A., & Cock, M. J. W. (2017). Classical Biological Control of *Mikania micrantha*: The Sustainable Solution. In C. A. Ellison, K. V. Sankaran, & S. T. Murphy (Eds.), *Invasive alien plants: Impacts on development and options for management.* (pp. 162–190). CABI. https://www.cabi.org/VetMedResource/ebook/20173322111

Ellison, C. A., & Sankaran, K. V. (2017). Profile of an invasive plant: *Mikania micrantha*. In C. A. Ellison, K. V. Sankaran, & S. T. Murphy (Eds.), *Invasive alien plants: Impacts on development and options for management.* (pp. 18–28). CABI. http://dx.doi.org/10.1079/9781780646275.0018

Elmqvist, T., Redman, C. L., Barthel, S., & Costanza, R. (2013). History of Urbanization and the Missing Ecology. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, S. Parnell, M. Schewenius, M. Sendstad, K. C. Seto, & C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 13–30). Springer. https://doi.org/10.1007/978-94-007-7088-1_2

El-Sayed, A. M., Suckling, D. M., Wearing, C. H., & Byers, J. A. (2006). Potential of Mass Trapping for Long-Term Pest Management and Eradication of Invasive Species. *Journal of Economic Entomology*, *99*(5), 1550–1564. https://doi.org/10.1603/0022-0493-99.5.1550

Ens, E., Daniels, C., Nelson, E., Roy, J., & Dixon, P. (2016). Creating multi-functional landscapes: Using exclusion fences to frame feral ungulate management preferences in remote Aboriginal-owned northern Australia. *Biological Conservation*, *197*, 235–246. https://doi.org/10.1016/j.biocon.2016.03.007

Epanchin-Niell, R. S., Brockerhoff, E. G., Kean, J. M., & Turner, J. A. (2014). Designing cost-efficient surveillance for early detection and control of multiple biological invaders. *Ecological Applications*, *24*(6), 1258–1274. https://doi.org/10.1890/13-1331.1

Epanchin-Niell, R. S., & Wilen, J. E. (2015). Individual and Cooperative Management of Invasive Species in Human-mediated Landscapes. *American Journal of Agricultural Economics*, *97*(1), 180–198. https://doi.org/10.1093/ajae/aau058

Epstein, G., & Smale, D. A. (2017). *Undaria pinnatifida*: A case study to highlight challenges in marine invasion ecology and management. *Ecology and Evolution*, *7*(20), 8624–8642. https://doi.org/10.1002/ece3.3430

Erz, W. (1966). Ecological principles in the urbanization of birds. *Ostrich*, *37*(sup1), 357–363. https://doi.org/10.1080/00306525.1966.9639812

Eschen, R., O'Hanlon, R., Santini, A., Vannini, A., Roques, A., Kirichenko, N., & Kenis, M. (2019). Safeguarding global plant health: The rise of sentinels. *Journal of Pest Science*, *92*(1), 29–36. https://doi.org/10.1007/s10340-018-1041-6

Espinosa-Romero, M. J., Chan, K. M. A., McDaniels, T., & Dalmer, D. M. (2011). Structuring decision-making for ecosystem-based management. *Marine Policy*, *35*(5), 575–583. https://doi.org/10.1016/j.marpol.2011.01.019

Essl, F., Bacher, S., Blackburn, T. M., Booy, O., Brundu, G., Brunel, S., Cardoso, A.-Cristina., Eschen, R., Gallardo, B., Galil, B., García-Berthou, E., Genovesi, P., Groom, Q., Harrower, C., Hulme, P. E., Katsanevakis, S., Kenis, M., Kühn, I., Kumschick, S., … Jeschke, J. M. (2015). Crossing Frontiers in Tackling Pathways of Biological Invasions. *BioScience*, *65*(8), 769–782. https://doi.org/10.1093/biosci/biv082

Essl, F., Nehring, S., Klingenstein, F., Milasowszky, N., Nowack, C., & Rabitsch, W. (2011). Review of risk assessment systems of IAS in Europe and introducing the German–Austrian Black List Information System (GABLIS). *Journal for Nature Conservation*, *19*(6), 339–350. https://doi.org/10.1016/j.jnc.2011.08.005

Estévez, R. A., Anderson, C. B., Pizarro, J. C., & Burgman, M. A. (2015). Clarifying values, risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species management. *Conservation Biology*, *29*(1), 19–30. https://doi.org/10.1111/cobi.12359

Estévez, R. A., Walshe, T., & Burgman, M. A. (2013). Capturing social impacts for decision-making: A Multicriteria Decision Analysis perspective. *Diversity and Distributions*, *19*(5–6), 608–616. https://doi.org/10.1111/ddi.12058

Esvelt, K. M., & Gemmell, N. J. (2017). Conservation demands safe gene drive. *PLOS Biology*, *15*(11), e2003850. https://doi.org/10.1371/journal.pbio.2003850

Etebari, K., Gharuka, M., Asgari, S., & Furlong, M. J. (2021). Diverse Host Immune Responses of Different Geographical Populations of the Coconut Rhinoceros Beetle to *Oryctes rhinoceros* Nudivirus (OrNV) Infection. *Microbiology Spectrum*, *9*(2), e00686-21. https://doi.org/10.1128/Spectrum.00686-21

European Union. (2014). *Regulation (EU) No 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species.* Official Journal of the European Union, 57, 35. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=OJ:JOL_2014_317_R_0003

FAO. (1995). *Guidelines for pest risk analysis*. International standards for phytosanitary measures (ISPM). No. 2. Food and Agriculture Organization of the United Nations. https://www.ippc.int/largefiles/adopted_ISPMs_previousversions/en/ISPM_02_1995_En_2006-05-03.pdf

FAO. (2006). *Recommendations for improved weed management*. FAO. https://www.fao.org/3/a0884e/a0884e00.pdf

FAO. (2008). *Understanding and applying risk analysis in aquaculture* (M. G. Bondad-Reantaso, J. R. Arthur, & R. P. Subasinghe, Eds.). FAO. https://www.fao.org/3/i0490e/i0490e00.pdf

FAO. (2018a). *International Standard for Phytosanitary Measures 6. Surveillance.* Food and Agriculture Organization of the United Nations. http://www.fao.org/documents/card/en/c/7985f320-a606-47f9-9f0b-9dfa5a2e1622/

FAO. (2018b). *The Republic of Namibia: Fall armyworm impact and needs assessment, 2018* (p. 52). Food and Agriculture Organisation of the United Nations. https://www.fao.org/documents/card/ru/c/I9556EN/

FAO. (2021). *Food and Agriculture Organization of the United Nations*. https://www.fao.org/food-chain-crisis/resources/success-stories/detail/en/c/1127653/

FAO & AEG. (2016). *Integrated Pest Management Using the Sterile Insect Technique*. Food and Agriculture Organization and Atomic Energy Agency. https://www.iaea.org/sites/default/files/sit-brochure.pdf

FAO & WHO. (2014). *The International Code of Conduct on Pesticide Management*. https://www.fao.org/fileadmin/templates/agphome/documents/Pests_Pesticides/Code/Code_ENG_2017updated.pdf

Farouk, A., & Zhen, D. (2019). Big data analysis techniques for intelligent systems. *Journal of Intelligent & Fuzzy Systems*, *37*(3), 3067–3071. https://doi.org/10.3233/JIFS-179109

Faulkner, K. T., Hulme, P. E., Pagad, S., Wilson, J. R. U., & Robertson, M. P. (2020). Classifying the introduction pathways of alien species: Are we moving in the right direction? *NeoBiota*, *62*, 143–159. https://doi.org/10.3897/neobiota.62.53543

Fenichel, E. P., Horan, R. D., & Bence, J. R. (2010). Indirect management of invasive species through bio-controls: A bioeconomic model of salmon and alewife in Lake Michigan. *Resource and Energy Economics*, *32*(4), 500–518. https://doi.org/10.1016/j.reseneeco.2010.04.002

Fenouillas, P., Ah-Peng, C., Amy, E., Bracco, I., Gosset, M., Ingrassia, F., Lavergne, C., Lequette, B., Notter, J. C., Pausé, J. M., Payet, N., Payet, G., Picot, F., Poungavanon, N., Strasberg, D., Thomas, H., Triolo, J., Turquet, V., & Rouget, V. (2020). *Priorisation spatiale des actions de gestion des plantes exotiques envahissantes: Une étape-clé de la conservation à long terme des milieux naturels à la Réunion. Version 1*. CIRAD, Saint Pierre. https://agritrop.cirad.fr/596376/1/Rapport%20technique%20PNR_24_Aout2020.pdf

Ferguson, A., & Chun, N. (2011). *Terminal Evaluation of Tonle Sap Conservation Project* (p. 60). Cambodia National Mekong Committee and Ministry of Environment, Royal Government of Cambodia. https://erc.undp.org/evaluation/documents/download/4959

Fernandes, J. A., Santos, L., Vance, T., Fileman, T., Smith, D., Bishop, J. D. D., Viard, F., Queirós, A. M., Merino, G., Buisman, E., & Austen, M. C. (2016). Costs and benefits to European shipping of ballast-water and hull-fouling treatment: Impacts of native and non-indigenous species. *Marine Policy*, *64*, 148–155. https://doi.org/10.1016/j.marpol.2015.11.015

Fessl, B., Young, G. H., Young, R. P., Rodríguez-Matamoros, J., Dvorak, M., Tebbich, S., & Fa, J. E. (2010). How to save the rarest Darwin's finch from extinction: The mangrove finch on Isabela Island. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1543), 1019–1030. https://doi.org/10.1098/rstb.2009.0288

Fields, P. G., & White, N. D. G. (2002). Alternatives to methyl bromide treatments for stored-product and quarantine insects. *Annual Review of Entomology*, *47*, 331–359. https://doi.org/10.1146/annurev.ento.47.091201.145217

Figueroa, D. F., McClure, A., Figueroa, N. J., & Hicks, D. W. (2019). Hiding in plain sight: Invasive coral *Tubastraea tagusensis* (Scleractinia: Hexacorallia) in the Gulf of Mexico. *Coral Reefs*, *38*(3), 395–403. https://doi.org/10.1007/s00338-019-01807-7

Filipe, J. A. N., Cobb, R. C., Meentemeyer, R. K., Lee, C. A., Valachovic, Y. S., Cook, A. R., Rizzo, D. M., & Gilligan, C. A. (2012). Landscape Epidemiology and Control of Pathogens with Cryptic and Long-Distance Dispersal: Sudden Oak Death in Northern Californian Forests. *PLOS Computational Biology*, *8*(1), e1002328. https://doi.org/10.1371/journal.pcbi.1002328

Finnoff, D., Shogren, J. F., Leung, B., & Lodge, D. (2007). Take a risk: Preferring prevention over control of biological invaders. *Ecological Economics*, *62*(2), 216–222. https://doi.org/10.1016/j.ecolecon.2006.03.025

Firn, J., House, A. P. N., & Buckley, Y. M. (2010). Alternative states models provide an effective framework for invasive species control and restoration of native communities. *Journal of Applied Ecology*, *47*(1), 96–105. https://doi.org/10.1111/j.1365-2664.2009.01741.x

Firn, J., Martin, T. G., Chadès, I., Walters, B., Hayes, J., Nicol, S., & Carwardine, J. (2015). Priority threat management of non-native plants to maintain ecosystem integrity across heterogeneous landscapes. *Journal of Applied Ecology*, *52*(5), 1135–1144. https://doi.org/10.1111/1365-2664.12500

Fish, J., Chiche, Y., Day, R., Efa, N., Witt, A., Fessehaie, R., De Graft Johnson, K., Gumisizira, G., & Nkandu, B. (2010). *Mainstreaming gender into prevention and management of invasive species*. Global Invasive Species Programme (GISP). https://portals.iucn.org/library/node/9837

Fleming, P. J. S., Allen, B. L., Allen, L. R., Ballard, G.-A., Bengsen, A., Gentle, M. N., McLeod, L. J., Meek, P. D., & Saunders, G. R. (2014). Management of wild canids in Australia: Free-ranging dogs and red foxes. In A. S. Glen & C. R. Dickman (Eds.), *Carnivores of Australia: Past, Present and Future* (pp. 105–149). CSIRO. https://scholar.google.com.au/scholar?hl=en&as_sdt=0%2C5&q=Management+of+wild+canids+in+Australia%3A+free-ranging+dogs+and+red+foxes.&btnG=

Fleming, P. J. S., Allen, L. R., Lapidge, S. J., Robley, A., Saunders, G. R., & Thomson, P. C. (2006). A strategic approach to mitigating the impacts of wild canids: Proposed activities of the Invasive Animals Cooperative Research Centre. *Australian Journal of Experimental Agriculture*, *46*(7), 753–762. https://doi.org/10.1071/EA06009

Flower, P. (2004). *Bitou Bush Helicopter Boom Spraying in NSW* (p. 96). http://byronshirechemicalfreelandcare.org/links/bitou-bush-helicopter-boom-spraying-in-nsw/

Flueck, W. T. (2010). The slippery slope of exporting invasive species: The case of Himalayan tahr arriving in South America. *Biological Invasions*, *12*(6), 1467–1475. https://doi.org/10.1007/s10530-009-9590-5

Follett, P. A., & Duan, J. J. (Eds.). (2000). *Nontarget effects of biological control*. Springer Science+Business Media New York. https://doi.org/10.1007/978-1-4615-4577-4

Fonseca, C. R., Guadagnin, D. L., Emer, C., Masciadri, S., Germain, P., & Zalba, S. M. (2013). Invasive alien plants in the Pampas grasslands: A tri-national cooperation challenge. *Biological Invasions*, *15*(8), 1751–1763. https://doi.org/10.1007/s10530-013-0406-2

Food Standard Agency. (2018). *Glossary of terms*. https://cot.food.gov.uk/moreinfo/cotglossary

Forbis, T. A., Provencher, L., Frid, L., & Medlyn, G. (2006). Great Basin Land Management Planning Using Ecological Modeling. *Environmental Management*, *38*(1), 62–83. https://doi.org/10.1007/s00267-005-0089-2

Forest Peoples Programme, the International Indigenous Forum on Biodiversity, & the Secretariat of the Convention on Biological Diversity. (2016). *Local Biodiversity Outlooks. Indigenous Peoples' and Local Communities' Contributions to the Implementation of the Strategic Plan for Biodiversity 2011-2020. A complement to the fourth edition of the Global Biodiversity Outlook*. https://www.cbd.int/gbo/gbo4/publication/lbo-en.pdf

Forrest, B. M., Hopkins, G. A., Dodgshun, T. J., & Gardner, J. P. A. (2007). Efficacy of acetic acid treatments in the management of marine biofouling. *Aquaculture*, *262*(2–4), 319–332. https://doi.org/10.1016/j.aquaculture.2006.11.006

Forsyth, D. M., Parkes, J. P., Woolnough, A. P., Pickles, G., Collins, M., & Gordon, I. (2009). Environmental and economic factors determine the number of feral goats commercially harvested in Western

Australia. *Journal of Applied Ecology*, *46*(1), 101–109. https://doi.org/10.1111/j.1365-2664.2008.01577.x

Forsyth, G. G., Le Maitre, D. C., O'Farrell, P. J., & van Wilgen, B. W. (2012). The prioritisation of invasive alien plant control projects using a multi-criteria decision model informed by stakeholder input and spatial data. *Journal of Environmental Management*, *103*, 51–57. https://doi.org/10.1016/j.jenvman.2012.01.034

Fowler, S. V. (2000). Trivial and political reasons for the failure of classical biological control of weeds: A personal view. *Proceedings of the X International Symposium on Biological Control of Weeds*, 169–172. http://bugwoodcloud.org/ibiocontrol/proceedings/pdf/10_169-172.pdf

Foxcroft, L. C., & McGeoch, M. (2011). Implementing invasive species management in an adaptive management framework. *Koedoe : African Protected Area Conservation and Science*, *53*(2), 1–11. https://doi.org/10.10520/EJC132237

Foxcroft, L. C., Pyšek, P., Richardson, D. M., & Genovesi, P. (Eds.). (2013). *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7

Foxcroft, L. C., Spear, D., van Wilgen, N. J., & McGeoch, M. A. (2019). Assessing the association between pathways of alien plant invaders and their impacts in protected areas. *NeoBiota*, *43*, 1–25. https://doi.org/10.3897/neobiota.43.29644

Foxcroft, L. C., Witt, A., & Lotter, W. D. (2013). Icons in peril: Invasive alien plants in African protected areas. In L. C. Foxcroft, P. Pyšek, P. Genovesi, & D. M. Richardson (Eds.), *Plant Invasions in Protected Areas. Patterns, Problems and Challenges.* (pp. 117–143). Springer. https://doi.org/10.1007/978-94-007-7750-7_7

Francis, R. A. (Ed.). (2011). *A Handbook of Global Freshwater Invasive Species*. Routledge. https://doi.org/10.4324/9780203127230

Frenot, Y., Chown, S. L., Whinam, J., Selkirk, P. M., Convey, P., Skotnicki, M., & Bergstrom, D. M. (2005). Biological invasions in the Antarctic: Extent, impacts and implications. *Biological Reviews*, *80*(1), 45–72. https://doi.org/10.1017/S1464793104006542

Friend, R. (2007). *Securing sustainable livelihoods through wise use of wetland resources: Reflections on the experience of the Mekong Wetlands Biodiversity Conservation and Sustainable Use Programme (MWBP)*. Mekong Wetlands Biodiversity Conservation and Sustainable Use Programme. https://portals.iucn.org/library/node/9198

Frost, R. A., & Launchbaugh, K. L. (2003). Prescription grazing for rangeland weed management. *Rangelands*, *25*(6), 43–47. http://dx.doi.org/10.2458/azu_rangelands_v25i6_frost

Froud, K. J., & Bullians, M. S. (2010). Investigation of biosecurity risk organisms for the plant and environment domains in New Zealand for 2008 and 2009. *New Zealand Plant Protection*, *63*, 262–269. https://doi.org/10.30843/nzpp.2010.63.6565

Furlan, E. M., Duncan, R., Bylemans, J., Hinlo, R., Rojahn, J., & Gleeson, D. (2018). Application of an eDNA detection framework for species monitoring. *Proceedings of Restore, Regenerate, Revegetate: A Conference on Restoring Ecological Processes, Ecosystems and Landscapes in a Changing World*, 33. https://hdl.handle.net/1959.11/22627

Furlan, E. M., Gleeson, D., Hardy, C. M., & Duncan, R. P. (2016). A framework for estimating the sensitivity of eDNA surveys. *Molecular Ecology Resources*, *16*(3), 641–654. https://doi.org/10.1111/1755-0998.12483

Furlan, E. M., Gleeson, D., Wisniewski, C., Yick, J., & Duncan, R. P. (2019). eDNA surveys to detect species at very low densities: A case study of European carp eradication in Tasmania, Australia. *Journal of Applied Ecology*, *56*(11), 2505–2517. https://doi.org/10.1111/1365-2664.13485

Gaertner, M., Larson, B. M. H., Irlich, U. M., Holmes, P. M., Stafford, L., van Wilgen, B. W., & Richardson, D. M. (2016). Managing invasive species in cities: A framework from Cape Town, South Africa. *Landscape and Urban Planning*, *151*, 1–9. https://doi.org/10.1016/j.landurbplan.2016.03.010

Gaertner, M., Wilson, J. R. U., Cadotte, M. W., MacIvor, J. S., Zenni, R. D., & Richardson, D. M. (2017). Non-native species in urban environments: Patterns, processes, impacts and challenges. *Biological Invasions*, *19*(12), 3461–3469. https://doi.org/10.1007/s10530-017-1598-7

Galil, B. S., Danovaro, R., Rothman, S. B. S., Gevili, R., & Goren, M. (2019). Invasive biota in the deep-sea Mediterranean: An emerging issue in marine conservation and management. *Biological Invasions*, *21*(2), 281–288. https://doi.org/10.1007/s10530-018-1826-9

Galil, B. S., McKenzie, C., Bailey, S. A., Campbell, M., Davidson, I. C., Drake, L., Hewitt, C., Occhipinti-Ambrogi, A., & Piola, R. (2019). *ICES Viewpoint background document: Evaluating and mitigating introduction of marine non-native species via vessel bio-fouling.* (p. 17) [ICES Ad Hoc Report 2019]. ICES. https://doi.org/10.17895/ices.pub.4680

Galil, B. S., Mienis, H. K., Hoffman, R., & Goren, M. (2021). Non-indigenous species along the Israeli Mediterranean coast: Tally, policy, outlook. *Hydrobiologia*, *848*(9), 2011–2029. https://doi.org/10.1007/s10750-020-04420-w

Gallardo, B., & Aldridge, D. C. (2013). The 'dirty dozen': Socio-economic factors amplify the invasion potential of 12 high-risk aquatic invasive species in Great Britain and Ireland. *Journal of Applied Ecology*, *50*(3), 757–766. https://doi.org/10.1111/1365-2664.12079

Gallien, L., Münkemüller, T., Albert, C. H., Boulangeat, I., & Thuiller, W. (2010). Predicting potential distributions of invasive species: Where to go from here? *Diversity and Distributions*, *16*(3), Article 3. https://doi.org/10.1111/j.1472-4642.2010.00652.x

Gallo, T., & Waitt, D. (2011). Creating a Successful Citizen Science Model to Detect and Report Invasive Species. *BioScience*, *61*(6), 459–465. https://doi.org/10.1525/bio.2011.61.6.8

Galt, R. E. (2010). Scaling Up Political Ecology: The Case of Illegal Pesticides on Fresh Vegetables Imported into the United States, 1996–2006. *Annals of the Association of American Geographers*, *100*(2), 327–355. https://doi.org/10.1080/00045601003595388

Gamage, T. V., Sanguansri, P., Swiergon, P., Eelkema, M., Wyatt, P., Leach, P., Alexander, D. L. J., & Knoerzer, K. (2015). Continuous combined microwave and hot air treatment of apples for fruit fly (*Bactrocera tryoni* and *B. jarvisi*) disinfestation. *Innovative Food Science & Emerging Technologies*, *29*, 261–270. https://doi.org/10.1016/j.ifset.2015.02.009

García Morales, M., Denno, B. D., Miller, D. R., Miller, G. L., Ben-Dov, Y., & Hardy, N. B. (2016). ScaleNet: A literature-based model of scale insect biology and systematics. *Database*, *2016*, bav118. https://doi.org/10.1093/database/bav118

Garcia-de-Lomas, J., & Vilà, M. (2015). Lists of harmful alien organisms: Are the national regulations adapted to the global world? *Biological Invasions*, *17*(11), 3081–3091. https://doi.org/10.1007/s10530-015-0939-7

García-Díaz, P., Ramsey, D. S. L., Woolnough, A. P., Franch, M., Llorente, G. A., Montori, A., Buenetxea, X., Larrinaga, A. R., Lasceve, M., Álvarez, A., Traverso, J. M., Valdeón, A., Crespo, A., Rada, V., Ayllón, E., Sancho, V., Lacomba, J. I., Bataller, J. V., & Lizana, M. (2017). Challenges in confirming eradication success of invasive red-eared sliders. *Biological Invasions*, *19*(9), 2739–2750. https://doi.org/10.1007/s10530-017-1480-7

García-Díaz, P., Ross, J. V., Woolnough, A. P., & Cassey, P. (2017). Managing the risk of wildlife disease introduction: Pathway-level biosecurity for preventing the introduction of alien ranaviruses. *Journal of Applied Ecology*, *54*(1), 234–241. https://doi.org/10.1111/1365-2664.12749

Gardener, M. R., Atkinson, R., & Rentería, J. L. (2010). Eradications and People: Lessons from the Plant Eradication Program in Galapagos. *Restoration Ecology*, *18*(1), 20–29. https://doi.org/10.1111/j.1526-100X.2009.00614.x

Gardener, M. R., Bustamante, R. O., Herrera, I., Durigan, G., Pivello, V. R., Moro, M. F., Stoll, A., Langdon, B., Baruch, Z., Rico, A., Arredondo-Nuñez, A., & Flores, S. (2012). Plant invasions research in Latin America: Fast track to a more focused agenda. *Plant Ecology & Diversity*, *5*(2), 225–232. https://doi.org/10.1080/17550874.2011.604800

Garner, M. G., & Beckett, S. D. (2005). Modelling the spread of foot-and-mouth disease in Australia. *Australian Veterinary Journal*, *83*(12), 758–766. https://doi.org/10.1111/j.1751-0813.2005.tb11589.x

Garnett, S. T., Burgess, N. D., Fa, J. E., Fernández-Llamazares, Á., Molnár, Z., Robinson, C. J., Watson, J. E. M., Zander, K. K., Austin, B., Brondizio, E. S., Collier, N. F., Duncan, T., Ellis, E., Geyle, H., Jackson, M. V., Jonas, H., Malmer, P., McGowan, B., Sivongxay, A., & Leiper, I. (2018). A spatial overview of the global importance of Indigenous lands for conservation. *Nature Sustainability*, *1*(7), 369–374. https://doi.org/10.1038/s41893-018-0100-6

Garrott, R. A., Siniff, D. B., Tester, J. R., Eagle, T. C., & Plotka, E. D. (1992). A Comparison of Contraceptive Technologies for Feral Horse Management. *Wildlife Society Bulletin (1973-2006)*, *20*(3), 318–326. https://www.jstor.org/stable/3783038

Garvey, P. M., Banks, P. B., Suraci, J. P., Bodey, T. W., Glen, A. S., Jones, C. J., McArthur, C., Norbury, G. L., Price, C. J., Russell, J. C., & Sih, A. (2020). Leveraging Motivations, Personality, and Sensory Cues for Vertebrate Pest Management. *Trends in Ecology & Evolution*, *35*(11), 990–1000. https://doi.org/10.1016/j.tree.2020.07.007

Gattani, A., Singh, S. V., Agrawal, A., Khan, M. H., & Singh, P. (2019). Recent progress in electrochemical biosensors as point of care diagnostics in livestock health. *Analytical Biochemistry*, *579*, 25–34. https://doi.org/10.1016/j.ab.2019.05.014

Gaudelli, N. M., Komor, A. C., Rees, H. A., Packer, M. S., Badran, A. H., Bryson, D. I., & Liu, D. R. (2017). Programmable base editing of A•T to G•C in genomic DNA without DNA cleavage. *Nature*, *551*(7681), 464–471. https://doi.org/10.1038/nature24644

Gebreziher, H. G., Gebreazgaabher, F. G., & Berhe, Y. K. (2021). Awareness creation of smallholder farmers on and adoption of push-pull technology reduces fall armyworm (*Spodoptera frugiperda*) infestation on maize in Hawzien Woreda, Northern Ethiopia. *Future of Food: Journal on Food, Agriculture and Society*, *9*(1), Article 1. https://doi.org/10.17170/KOBRA-202011192210

Geering, W. A., & Lubroth, J. (2002). *Preparation of foot-and-mouth disease contingency plans*. Food and Agricultural Organization of the United Nations.

GEF. (2007). *Mainstreaming Prevention and Control Measures for Invasive Alien Species into Trade, Transport and Travel Across the Production Landscape*. Global Environment Facility. https://www.thegef.org/project/mainstreaming-prevention-and-control-measures-invasive-alien-species-trade-transport-and

GEF. (2020). *South Pacific Biodiversity Conservation Programme*. Global Environment Facility. https://www.thegef.org/projects-operations/projects/403

Genovesi, P. (2001). *Guidelines for Eradication of Terrestrial Vertebrates: A European Contribution to the Invasive Alien Species Issue* (Other Publications in Wildlife Management, p. 29). https://digitalcommons.unl.edu/cgi/viewcontent.cgi?referer=&httpsredir=1&article=1023&context=icwdmother

Genovesi, P. (2011). Are we turning the tide? Eradications in times of crisis: How the global community is responding to biological invasions. In C. R. Veitch, M. N. Clout, & D. R. Towns (Eds.), *Island Invasives: Eradication and Management. Proceedings of the International Conference on Island Invasives* (p. 4). IUCN and The Centre for Biodiversity and Biosecurity (CBB). https://www.researchgate.net/publication/233755935_Are_we_turning_the_tide_Eradications_in_times_of_crisis_how_the_global_community_is_responding_to_biological_invasions

Genovesi, P., & Bertolino, S. (2001). Human dimension aspects in invasive alien species issues: The case of the failure of the grey squirrel eradication project in Italy. In J. A. McNeely (Ed.), *The Great Reshuffling: Human Dimensions of Invasive Alien Species* (pp. 113–119). IUCN – The World Conservation Union. https://www.researchgate.net/publication/233809634_Human_dimension_aspects_in_invasive_alien_species_issues_the_case_of_the_failure_of_the_grey_squirrel_eradication_project_in_Italy

Gentle, M., & Pople, A. (2013). Effectiveness of commercial harvesting in controlling feral-pig populations. *Wildlife Research*, *40*(6), 459–469. https://doi.org/10.1071/WR13100

George, J., Häsler, B., Komba, E., Sindato, C., Rweyemamu, M., & Mlangwa, J. (2021). Towards an integrated animal health surveillance system in Tanzania: Making better use of existing and potential data sources for early warning surveillance. *BMC Veterinary Research*, *17*(1), 109. https://doi.org/10.1186/s12917-021-02789-x

Gerda. (2021). *Gerda · global eradication and response database*. Global Eradication and Response Database. http://b3.net.nz/gerda/

Giachetti, C. B., Battini, N., Castro, K. L., & Schwindt, E. (2020). Invasive ascidians: How predators reduce their dominance in artificial structures in cold temperate areas. *Journal of Experimental Marine Biology and Ecology*, *533*, 151459. https://doi.org/10.1016/j.jembe.2020.151459

Giakoumi, S., Katsanevakis, S., Albano, P. G., Azzurro, E., Cardoso, A. C., Cebrian, E., Deidun, A., Edelist, D., Francour, P., Jimenez, C., Mačić, V., Occhipinti-Ambrogi, A., Rilov, G., & Sghaier, Y. R. (2019). Management priorities for marine invasive species. *Science of The Total Environment*, *688*, 976–982. https://doi.org/10.1016/j.scitotenv.2019.06.282

Giakoumi, S., Pey, A., Di Franco, A., Francour, P., Kizilkaya, Z., Arda, Y., Raybaud, V., & Guidetti, P. (2019). Exploring the relationships between marine protected areas and invasive fish in the world's most invaded sea. *Ecological Applications*, *29*(1), e01809. https://doi.org/10.1002/eap.1809

Gierus, L., Birand, A., Bunting, M. D., Godahewa, G. I., Piltz, S. G., Oh, K. P., Piaggio, A. J., Threadgill, D. W., Godwin, J., Edwards, O., Cassey, P., Ross, J. V., Prowse, T. A. A., & Thomas, P. Q. (2022). Leveraging a natural murine meiotic drive to suppress invasive populations. *Proceedings of the National Academy of Sciences*, *119*(46), e2213308119. https://doi.org/10.1073/pnas.2213308119

Gigon, A. (2007). *Ersatz-Pflanzenarten für die unerwünschten gebietsfremden Arten (invasive Neophyten) der Schwarzen und der Beobachtungsliste der Schweiz* (p. 12). https://www.infoflora.ch/de/assets/content/documents/neophyten/neophyten_diverses/neophyten-ersatzarten.pdf

Girsang, S. S., Nurzannah, S. E., Girsang, M. A., & Effendi, R. (2020). The distribution and impact of fall army worm (*Spodoptera frugiperda*) on maize production in North Sumatera. *IOP Conference Series: Earth and Environmental Science*, *484*(1), 012099. https://doi.org/10.1088/1755-1315/484/1/012099

Glare, T., Caradus, J., Gelernter, W., Jackson, T., Keyhani, N., Köhl, J., Marrone, P., Morin, L., & Stewart, A. (2012). Have biopesticides come of age? *Trends in Biotechnology*, *30*(5), 250–258. https://doi.org/10.1016/j.tibtech.2012.01.003

Glen, A. S., Atkinson, R., Campbell, K. J., Hagen, E., Holmes, N. D., Keitt, B. S., Parkes, J. P., Saunders, A., Sawyer, J., & Torres, H. (2013). Eradicating multiple invasive species on inhabited islands: The next big step in island restoration? *Biological Invasions*, *15*(12), 2589–2603. https://doi.org/10.1007/s10530-013-0495-y

Glen, A. S., Latham, M. C., Anderson, D., Leckie, C., Niemiec, R., Pech, R. P., & Byrom, A. E. (2017). Landholder participation in regional-scale control of invasive predators: An adaptable landscape model. *Biological Invasions*, *19*(1), 329–338. https://doi.org/10.1007/s10530-016-1282-3

Godfree, R. C., Knerr, N., Godfree, D., Busby, J., Robertson, B., & Encinas-Viso, F. (2019). Historical reconstruction unveils the risk of mass mortality and ecosystem collapse during pancontinental megadrought. *Proceedings of the National Academy of Sciences*, *116*(31), 15580–15589. https://doi.org/10.1073/pnas.1902046116

Goes, F. (2005). *Four years of water bird conservation activities in Prek Toal Core Area, Tonle Sap Biosphere Reserve (2001-2004)*. Wildlife Conservation Society. https://library.khmerstudies.org/bib/2583

Goka, K. (2010). Introduction to the Special Feature for Ecological Risk Assessment of Introduced Bumblebees: Status of the European bumblebee, *Bombus terrestris*, in Japan as a beneficial pollinator and an invasive alien species. *Applied Entomology and Zoology*, *45*(1), 1–6. https://doi.org/10.1303/aez.2010.1

Göktoćan, A. H., Sukkarieh, S., Bryson, M., Randle, J., Lupton, T., & Hung, C. (2010). A Rotary-wing Unmanned Air Vehicle for Aquatic Weed Surveillance and Management. *Journal of Intelligent and Robotic Systems*, *57*(1), 467–484. https://doi.org/10.1007/s10846-009-9371-5

Gollasch, S., & Nehring, S. (2006). National checklist for aquatic alien species in Germany. *Aquatic Invasions*, *1*(4), 245–269. https://doi.org/10.3391/ai.2006.1.4.8

Gonzalez, L. F., Montes, G. A., Puig, E., Johnson, S., Mengersen, K., & Gaston, K. J. (2016). Unmanned Aerial Vehicles (UAVs) and Artificial Intelligence Revolutionizing Wildlife Monitoring and Conservation. *Sensors*, *16*(1), 97. https://doi.org/10.3390/s16010097

González-Moreno, P., Lazzaro, L., Vilà, M., Preda, C., Adriaens, T., Bacher, S., Brundu, G., Copp, G. H., Essl, F., García-Berthou, E., Katsanevakis, S., Moen, T. L., Lucy, F. E., Nentwig, W., Roy, H. E., Srėbalienė, G., Talgø, V., Vanderhoeven, S., Andjelković, A., … Kenis, M. (2019). Consistency of impact assessment protocols for non-native species. *NeoBiota*, *44*, 1–25. https://doi.org/10.3897/neobiota.44.31650

Gopal, P., Bordoloi, S., Ratnam, R., Lin, P., Cai, W., Buragohain, P., Garg, A., & Sreedeep, S. (2019). Investigation of infiltration rate for soil-biochar composites of water hyacinth. *Acta Geophysica*, *67*(1), 231–246. https://doi.org/10.1007/s11600-018-0237-8

Gosling, L. M., & Baker, S. J. (1989). The eradication of muskrats and coypus from Britain. *Biological Journal of the Linnean Society*, *38*(1), 39–51. https://doi.org/10.1111/j.1095-8312.1989.tb01561.x

Gozlan, R. E., Andreou, D., Asaeda, T., Beyer, K., Bouhadad, R., Burnard, D., Caiola, N., Cakic, P., Djikanovic, V., Esmaeili, H. R., Falka, I., Golicher, D., Harka, A., Jeney, G., Kováč, V., Musil, J., Nocita, A., Povz, M., Poulet, N., … Britton, J. R. (2010). Pan-continental invasion of *Pseudorasbora parva*: Towards a better understanding of freshwater fish invasions. *Fish and Fisheries*, *11*(4), 315–340. https://doi.org/10.1111/j.1467-2979.2010.00361.x

Grace, K. E. F., Papadopoulou, C., Floyd, T., Avigad, R., Collins, S., White, E., Batten, C., Flannery, J., Gubbins, S., & Carpenter, S. T. (2020). Risk-based surveillance for bluetongue virus in cattle on the south coast of England in 2017 and 2018. *Veterinary Record*, *187*(11), e96. https://doi.org/10.1136/vr.106016

Graham, K., Gilligan, D., Brown, P., van Klinken, R. D., McColl, K. A., & Durr, P. A. (2021). Use of spatio-temporal habitat suitability modelling to prioritise areas for common carp biocontrol in Australia using the virus CyHV-3. *Journal of Environmental Management*, *295*, 113061. https://doi.org/10.1016/j.jenvman.2021.113061

Graham, S., Metcalf, A. L., Gill, N., Niemiec, R., Moreno, C., Bach, T., Ikutegbe, V., Hallstrom, L., Ma, Z., & Lubeck, A. (2019). Opportunities for better use of collective action theory in research and governance for invasive species management. *Conservation Biology*, *33*(2), 275–287. https://doi.org/10.1111/cobi.13266

Grechi, I., Chadès, I., Buckley, Y. M., Friedel, M. H., Grice, A. C., Possingham, H. P., van Klinken, R. D., & Martin, T. G. (2014). A decision framework for management of conflicting production and biodiversity goals for a commercially valuable invasive species. *Agricultural Systems*, *125*, 1–11. https://doi.org/10.1016/j.agsy.2013.11.005

Green, S. J., Underwood, E. B., & Akins, J. L. (2017). Mobilizing volunteers to sustain local suppression of a global marine invasion. *Conservation Letters*, *10*(6), 726–735. https://doi.org/10.1111/conl.12426

Gregory, R., Failing, L., Ohlson, D., & Mcdaniels, T. L. (2006). Some Pitfalls of an Overemphasis on Science in Environmental Risk Management Decisions. *Journal of Risk Research*, *9*(7), 717–735. https://doi.org/10.1080/13669870600799895

Gregory, S., Henderson, W., Smee, E., & Cassey, P. (2014). *Eradications of vertebrate pests in Australia: A review and guidelines for future best practice*. Canberra, Australia: Invasive Animals Cooperative Research Centre. https://pestsmart.org.au/wp-content/uploads/sites/3/2020/06/AusEradications_2014.pdf

Grice, A. C. (2006). Commercially valuable weeds: Can we eat our cake without choking on it? *Ecological Management & Restoration*, *7*(1), 40–44. https://doi.org/10.1111/j.1442-8903.2006.00246.x

Grice, A. C., Clarkson, J. D., Murphy, H. T., Fletcher, C. S., & Westcott, D. A. (2013). Containment as a strategic option for managing plant invasion. *Plant Protection Quarterly*, *28*(3), 62–65. https://search.informit.org/doi/10.3316/ielapa.685341871862730

Grice, A. C., Clarkson, J. R., Friedel, M. H., Murphy, H. T., Fletcher, C. S., & Westcott, D. A. (2012). Containment: The state of play. *Developing Solutions to Evolving Weed Problems. 18th Australasian Weeds Conference*, 320–324. https://caws.org.nz/old-site/awc/2012/awc201213201.pdf

Grice, A. C., Murphy, H. T., Clarkson, J. R., Friedel, M. H., Fletcher, C. S., & Westcott, D. A. (2020). A review and refinement of the concept of containment for the management of invasive plants. *Australian Journal of Botany*, *68*(8), 602–616. https://doi.org/10.1071/BT20092

Griffiths, R., Cranwell, S., Derand, D., Ghestemme, T., Will, D., Zito, J., Hall, T., Pott, M., & Coulston, G. (2019). Multi island, multi invasive species eradication in French Polynesia demonstrates economies of scale. In C. R. Veitch, M. N. Clout, A. R. Martin, J. C. Russell, & C. J. West (Eds.), *Island invasives: Scaling up to meet the challenge* (pp. 611–617). IUCN SSC Invasive Species Specialist Group. http://www.issg.org/pdf/publications/Island_Invasives/pdfHQprint/2Griffiths.pdf

Groom, Q. J., Desmet, P., Reyserhove, L., Adriaens, T., Oldoni, D., Vanderhoeven, S., Baskauf, S. J., Chapman, A., McGeoch, M. A., Walls, R., Wieczorek, J., Wilson, J. R. U., Zermoglio, P. F., & Simpson, A. (2019). Improving Darwin Core for research and management of alien species. *Biodiversity Information Science and Standards*, *3*, e38084. https://doi.org/10.3897/biss.3.38084

Groom, Q. J., Strubbe, D., Adriaens, T., Davis, A. J. S., Desmet, P., Oldoni, D., Reyserhove, L., Roy, H. E., & Vanderhoeven, S. (2019). Empowering Citizens to Inform Decision-Making as a Way Forward to Support Invasive Alien Species Policy. *Citizen Science: Theory and Practice*, *4*(1), 33. https://doi.org/10.5334/cstp.238

Grosholz, E., Ashton, G., Bradley, M., Brown, C., Ceballos-Osuna, L., Chang, A., de Rivera, C., Gonzalez, J., Heineke, M., Marraffini, M., McCann, L., Pollard, E., Pritchard, I., Ruiz, G., Turner, B., & Tepolt, C. (2021). Stage-specific overcompensation, the hydra effect, and the failure to eradicate an invasive predator. *Proceedings of the National Academy of Sciences*, *118*(12), e2003955118. https://doi.org/10.1073/pnas.2003955118

Grossel, G., Lyon, A., & Nunn, M. (2017). Open-Source Intelligence Gathering and Open-Analysis Intelligence for Biosecurity. In A. P. Robinson, T. Walshe, M. A. Burgman, & M. Nunn (Eds.), *Invasive Species: Risk Assessment and Management* (1st ed., pp. 84–92). Cambridge University Press. https://doi.org/10.1017/9781139019606.005

Groves, R. H., Panetta, F. D., & Virtue, J. G. (Eds.). (2001). *Weed Risk Assessment*. CSIRO Publishing.

Gruber, M. A. M., Cooling, M., Baty, J. W., Buckley, K., Friedlander, A., Quinn, O., Russell, J. F. E. J., Sébastien, A., & Lester, P. J. (2017). Single-stranded RNA viruses infecting the invasive Argentine ant, *Linepithema humile*. *Scientific Reports*, *7*(1), 3304. https://doi.org/10.1038/s41598-017-03508-z

Gruber, M. A. M., Cooling, M., & Burne, A. R. (2016). *PIAT Pacific Invasive Ant toolkit. Pacific Biosecurity. New Zealand Ministry of Foreign Affairs and Trade.* https://piat.org.nz/index.php?page=citing-the-toolkit

Grung, M., Lin, Y., Zhang, H., Steen, A. O., Huang, J., Zhang, G., & Larssen, T. (2015). Pesticide levels and environmental risk in aquatic environments in China—A review. *Environment International*, *81*, 87–97. https://doi.org/10.1016/j.envint.2015.04.013

Gu, D. E., Yu, F. D., Yang, Y. X., Xu, M., Wei, H., Luo, D., Mu, X. D., & Hu, Y. C. (2019). Tilapia fisheries in Guangdong Province, China: Socio-economic benefits, and threats on native ecosystems and economics. *Fisheries Management and Ecology*, *26*(2), 97–107. https://doi.org/10.1111/fme.12330

Gu, D. E., Yu, F.-D., Yang, Y.-X., Ma, G.-M., Hu, Y.-C., Mu, X.-D., Xu, M., Liu, C., Luo, D., & Wei, H. (2018). Killing effect of "miefeiling" on alien nile tilapia in ornamental aquaculture. *Chinese Journal of Ecology*, *37*(10), 2985–2988. https://doi.org/10.13292/j.1000-4890.201810.001

Guareschi, S., & Wood, P. J. (2019). Taxonomic changes and non-native species: An overview of constraints and new challenges for macroinvertebrate-based indices calculation in river ecosystems. *Science of The Total Environment*, *660*, 40–46. https://doi.org/10.1016/j.scitotenv.2019.01.008

Gupta, A. D. (2015). Invasive Pest Attack in North Bengal. In E. Ens, J. Fisher, & O. Costello (Eds.), *Indigenous people and invasive species: Perceptions, management, challenges and uses. IUCN Commission on Ecosystem Management Community Report.* https://ipm.ifas.ufl.edu/pdfs/ens_et_al_2015_indigenous_people_and_invasive_species_iucn_cem_ecosystems_and_invasiv.pdf

Gupta, K., & Sankaran, K. V. (2021). Forest Biosecurity Systems and Processes: An Indian Perspective. *Frontiers in Forests and Global Change*, *4*, 699950. https://doi.org/10.3389/ffgc.2021.699950

Gürtler, R. E., Izquierdo, V. M., Gil, G., Cavicchia, M., & Maranta, A. (2017). Coping with wild boar in a conservation area: Impacts of a 10-year management control program in north-eastern Argentina. *Biological Invasions*, *19*, 11–24. https://doi.org/10.1007/s10530-016-1256-5

Gürtler, R. E., Rodríguez-Planes, L., Gil, G., Izquierdo, V. M., Cavicchia, M., & Maranta, A. (2018). Differential long-term impacts of a management control program of axis deer and wild boar in a protected area of north-eastern Argentina. *Biological Invasions*, *20*, 1431–1447. https://doi.org/10.1007/s10530-017-1635-6

Gutierrez, J. B., & Teem, J. L. (2006). A model describing the effect of sex-reversed YY fish in an established wild population: The use of a Trojan Y chromosome to cause extinction of an introduced exotic species. *Journal of Theoretical Biology*, *241*(2), 333–341. https://doi.org/10.1016/j.jtbi.2005.11.032

Haack, R. A., Britton, K. O., Brockerhoff, E. G., Cavey, J. F., Garrett, L. J., Kimberley, M., Lowenstein, F., Nuding, A., Olson, L. J., Turner, J., & Vasilaky, K. N. (2014). Effectiveness of the International Phytosanitary Standard ISPM No. 15 on Reducing Wood Borer Infestation Rates in Wood Packaging Material Entering the United States. *PLoS ONE*, *9*(5), e96611. https://doi.org/10.1371/journal.pone.0096611

Hajek, A. E., McManus, M. L., & Delalibera, I. (2007). A review of introductions of pathogens and nematodes for classical biological control of insects and mites. *Biological Control*, *41*(1), 1–13. https://doi.org/10.1016/j.biocontrol.2006.11.003

Hall, S. A., Bastos, R., Vicente, J., Vaz, A. S., Honrado, J. P., Holmes, P. M., Gaertner, M., Esler, K. J., & Cabral, J. A. (2021). A dynamic modeling tool to anticipate the effectiveness of invasive plant control and restoration recovery trajectories in South

African fynbos. *Restoration Ecology*, *29*(3), e13324. https://doi.org/10.1111/rec.13324

Handford, C. E., Elliott, C. T., & Campbell, K. (2015). A review of the global pesticide legislation and the scale of challenge in reaching the global harmonization of food safety standards. *Integrated Environmental Assessment and Management*, *11*(4), 525–536. https://doi.org/10.1002/ieam.1635

Hänfling, B., Edwards, F., & Gherardi, F. (2011). Invasive alien Crustacea: Dispersal, establishment, impact and control. *BioControl*, *56*(4), 573–595. https://doi.org/10.1007/s10526-011-9380-8

Hansen, F. T., Potthoff, M., Uhrenholdt, T., Vo, H. D., Linden, O., & Andersen, J. H. (2015). Development of a prototype tool for ballast water risk management using a combination of hydrodynamic models and agent-based modeling. *WMU Journal of Maritime Affairs*, *14*(2), 219–245. https://doi.org/10.1007/s13437-014-0067-8

Haregeweyn, N., Tsunekawa, A., Tsubo, M., Meshesha, D., & Melkie, A. (2013). Analysis of the invasion rate, impacts and control measures of *Prosopis juliflora*: A case study of Amibara District, Eastern Ethiopia. *Environmental Monitoring and Assessment*, *185*(9), 7527–7542. https://doi.org/10.1007/s10661-013-3117-3

Harrington, R. A., Kujawski, R., & Ryan, H. D. P. (2003). Invasive plants and the green industry. *Journal of Arboriculture*, *29*(1), 42–48. https://joa.isa-arbor.com/request.asp?JournalID=1&ArticleID=70&Type=2

Harrington, R. A., Taylor, M. S., Shortall, C. R., Alderson, L., Mallott, M., & Verrier, P. J. (2012). The Rothamsted Insect Survey: Old traps, new tricks. *Aspects of Applied Biology*, *117*, 157–164. https://repository.rothamsted.ac.uk/item/8qv68/the-rothamsted-insect-survey-old-traps-new-tricks

Harris, D. B., Gregory, S. D., Bull, L. S., & Courchamp, F. (2012). Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk. *Biological Invasions*, *14*(6), 1251–1263. https://doi.org/10.1007/s10530-011-0153-1

Harwood, T. D., Tomlinson, I., Potter, C. A., & Knight, J. D. (2011). Dutch elm disease revisited: Past, present and future management in Great Britain. *Plant Pathology*, *60*(3), 545–555. https://doi.org/10.1111/j.1365-3059.2010.02391.x

Hassell, M. P. (1978). *The Dynamics of Arthopod Predator-Prey Systems. (MPB-13), Volume 13*. Princeton University Press.

Hatcher, P. E., & Melander, B. (2003). Combining physical, cultural and biological methods: Prospects for integrated non-chemical weed management strategies. *Weed Research*, *43*(5), 303–322. https://doi.org/10.1046/j.1365-3180.2003.00352.x

Hay, S. I., George, D. B., Moyes, C. L., & Brownstein, J. S. (2013). Big Data Opportunities for Global Infectious Disease Surveillance. *PLOS Medicine*, *10*(4), e1001413. https://doi.org/10.1371/journal.pmed.1001413

Haye, T., Olfert, O., Weiss, R., Mason, P. G., Gibson, G., Gariepy, T. D., & Gillespie, D. R. (2018). Bioclimatic analyses of *Trichomalus perfectus* and *Mesopolobus morys* (Hymenoptera: Pteromalidae) distributions, two potential biological control agents of the cabbage seedpod weevil in North America. *Biological Control*, *124*, 30–39. https://doi.org/10.1016/j.biocontrol.2018.06.003

Hayes, K. R., Hosack, G. R., Dana, G. V., Foster, S. D., Ford, J. H., Thresher, R., Ickowicz, A., Peel, D., Tizard, M., De Barro, P., Strive, T., & Dambacher, J. M. (2018). Identifying and detecting potentially adverse ecological outcomes associated with the release of gene-drive modified organisms. *Journal of Responsible Innovation*, *5*(sup1), S139–S158. https://doi.org/10.1080/23299460.2017.1415585

Hayes, K. R., Hosack, G. R., Ickowicz, A., Foster, S., Peel, D., Ford, J., & Thresher, R. (2018). *Risk Assessment for Controlling Mosquito Vectors with Engineered Nucleases: Contained field release for Sterile Male Construct Risk assessment final report*. CSIRO. https://doi.org/10.25919/v424-4k80

Hayes, K. R., Inglis, G. J., & Barry, S. C. (2019). The Assessment and Management of Marine Pest Risks Posed by Shipping: The Australian and New Zealand Experience. *Frontiers in Marine Science*, *6*, 489. https://doi.org/10.3389/fmars.2019.00489

Hayes, R. A., Griffiths, M. W., Nahrung, H. F., Arnold, P. A., Hanks, L. M., & Millar, J. G. (2016). Optimizing Generic Cerambycid Pheromone Lures for Australian Biosecurity and Biodiversity Monitoring. *Journal of Economic Entomology*, *109*(4), 1741–1749. https://doi.org/10.1093/jee/tow100

Hazelton, E. L. G., Mozdzer, T. J., Burdick, D. M., Kettenring, K. M., & Whigham, D. F. (2014). *Phragmites australis* management in the United States: 40 years of methods and outcomes. *AoB PLANTS*, *6*, plu001. https://doi.org/10.1093/aobpla/plu001

Hedley, J. (2005). The International Plant Protection Convention and Invasive Species. *Environmental Law Reporter News & Analysis*, *35*, 10381. https://heinonline.org/HOL/Page?handle=hein.journals/elrna35&id=344&div=&collection=

Heikkilä, J. (2011). A review of risk prioritisation schemes of pathogens, pests and weeds: Principles and practices. *Agricultural and Food Science*, *20*(1), 15–28. https://doi.org/10.2137/145960611795163088

Heimpel, G. E., & Mills, N. J. (2017). *Biological Control*. Cambridge University Press.

Helinski, M. E. H., & Knols, B. G. J. (2008). Mating Competitiveness of Male *Anopheles arabiensis* Mosquitoes Irradiated with a Partially or Fully Sterilizing Dose in Small and Large Laboratory Cages. *Journal of Medical Entomology*, *45*(4), 698–705. https://doi.org/10.1093/jmedent/45.4.698

Hellmann, J. J., Byers, J. E., Bierwagen, B. G., & Dukes, J. S. (2008). Five Potential Consequences of Climate Change for Invasive Species. *Conservation Biology*, *22*(3), 534–543. https://doi.org/10.1111/j.1523-1739.2008.00951.x

Helmstedt, K. J., Shaw, J. D., Bode, M., Terauds, A., Springer, K., Robinson, S. A., & Possingham, H. P. (2016). Prioritizing eradication actions on islands: It's not all or nothing. *Journal of Applied Ecology*, *53*(3), 733–741. https://doi.org/10.1111/1365-2664.12599

Herder, J. E., Valentini, A., Bellemain, E., Dejean, T., van Delft, J. J. C. W., Thomsen, P. F., & Taberlet, P. (2014). *Environmental DNA – a review of the possible applications for the detection of (invasive) species* (Report 2013-104). Netherlands Food and Consumer Product Safety Authority. https://doi.org/10.13140/RG.2.1.4002.1208

Herms, D. A., & McCullough, D. G. (2014). Emerald Ash Borer Invasion of North America: History, Biology, Ecology, Impacts, and Management. *Annual Review of Entomology*, *59*(1), 13–30. https://doi.org/10.1146/annurev-ento-011613-162051

Herring, C. E., Stinson, J., & Landis, W. G. (2015). Evaluating nonindigenous species management in a Bayesian networks derived relative risk framework for Padilla Bay, WA, USA: Risk Assessment for Nonindigenous Species. *Integrated Environmental Assessment and Management*, *11*(4), 640–652. https://doi.org/10.1002/ieam.1643

Hershenhorn, J., Casella, F., & Vurro, M. (2016). Weed biocontrol with fungi: Past, present and future. *Biocontrol Science and Technology*, *26*(10), 1313–1328. https://doi.org/10.1080/09583157.2016.1209161

Heuzé, V., Tran, G., Hassoun, P., Régnier, C., Bastianelli, D., & Lebas, F. (2015). *Water hyacinth (*Eichhornia crassipes*). Feedipedia, a programme by INRA, CIRAD, AFZ and FAO.* https://www.feedipedia.org/node/160 Last updated on October 13, 2015, 16:25

Hewitt, C. L., & Campbell, M. L. (2007). Mechanisms for the prevention of marine bioinvasions for better biosecurity. *Marine Pollution Bulletin*, *55*(7–9), 395–401. https://doi.org/10.1016/j.marpolbul.2007.01.005

Hewitt, C. L., & Campbell, M. L. (2010). *The relative contribution of vectors to the introduction and translocation of invasive marine species*. The Department of Agriculture, Fisheries and Forestry (DAFF). https://www.marinepests.gov.au/sites/default/files/Documents/relative-contribution-vectors-introduction-translocation-invasive-marine-species.pdf

Hewitt, C. L., Willing, J., Bauckham, A., Cassidy, A. M., Cox, C. M. S., Jones, L., & Wotton, D. M. (2004). New Zealand marine biosecurity: Delivering outcomes in a fluid environment. *New Zealand Journal of Marine and Freshwater Research*, *38*(3), 429–438. https://doi.org/10.1080/00288330.2004.9517250

Heywood, V., & Brunel, S. (2011). Code of conduct on horticulture and invasive alien plants. *Italian Botanist*, *44*, 3–47. https://www.scopus.com/inward/record.uri?eid=2-s2.0-85050005463&partnerID=40&md5=e4dc4ea7c9bf290cf2817c4520fada4d

Hiebert, R. D., & Stubbendieck, J. (1993). *Handbook for ranking exotic plants for management and control* (Natural Resources Report NPS/NRMWRO/NRR-93/08). U. S. Department of the Interior, National Park Service. Natural Resources Publication Office Denver, Colorado. https://agris.fao.org/agris-search/search.do?recordID=US201300166280

Hieda, S., Kaneko, Y., Nakagawa, M., & Noma, N. (2020). *Ludwigia grandiflora* (Michx.) Greuter & Burdet subsp. *Hexapetala* (Hook. & Arn.) G. L. Nesom & Kartesz, an Invasive Aquatic Plant in Lake Biwa, the Largest Lake in Japan. *Acta Phytotaxonomica et Geobotanica*, *71*(1), 65–71. https://doi.org/10.18942/apg.201911

Hill, D. J., Tarasoff, C., Whitworth, G. E., Baron, J., Bradshaw, J. L., & Church, J. S. (2017). Utility of unmanned aerial vehicles for mapping invasive plant species: A case study on yellow flag iris (*Iris pseudacorus* L.). *International Journal of Remote Sensing*, *38*(8–10), 2083–2105. https://doi.org/10.1080/01431161.2016.1264030

Hinlo, R., Furlan, E., Suitor, L., & Gleeson, D. (2017). Environmental DNA monitoring and management of invasive fish: Comparison of eDNA and fyke netting. *Management of Biological Invasions*, *8*(1), 89–100. https://doi.org/10.3391/mbi.2017.8.1.09

Hoagland, P., & Jin, D. (2006). Science and Economics in the Management of an Invasive Species. *BioScience*, *56*(11), 931–935. https://doi.org/10.1641/0006-3568(2006)56[931:SAEITM]2.0.CO;2

Hobbs, N. T., Bowden, D. C., & Baker, D. L. (2000). Effects of Fertility Control on Populations of Ungulates: General, Stage-Structured Models. *The Journal of Wildlife Management*, *64*(2), 473–491. JSTOR. https://doi.org/10.2307/3803245

Hoffmann, B. D., Luque, G. M., Bellard, C., Holmes, N. D., & Donlan, C. J. (2016). Improving invasive ant eradication as a conservation tool: A review. *Biological Conservation*, *198*, 37–49. https://doi.org/10.1016/j.biocon.2016.03.036

Hoinville, L. J., Alban, L., Drewe, J. A., Gibbens, J. C., Gustafson, L., Häsler, B., Saegerman, C., Salman, M., & Stärk, K. D. C. (2013). Proposed terms and concepts for describing and evaluating animal-health surveillance systems. *Preventive Veterinary Medicine*, *112*(1–2), 1–12. https://doi.org/10.1016/j.prevetmed.2013.06.006

Holbrook, F. R., & Fujimoto, M. S. (1970). Mating Competitiveness of Unirradiated and Irradiated Mediterranean Fruit Flies. *Journal of Economic Entomology*, *63*(4), 1175–1176. https://doi.org/10.1093/jee/63.4.1175

Holmala, K., & Kauhala, K. (2006). Ecology of wildlife rabies in Europe. *Mammal Review*, *36*(1), 17–36. https://doi.org/10.1111/j.1365-2907.2006.00078.x

Holman, L. E., de Bruyn, M., Creer, S., Carvalho, G., Robidart, J., & Rius, M. (2019). Detection of introduced and resident marine species using environmental DNA metabarcoding of sediment and water. *Scientific Reports*, *9*(1), 11559. https://doi.org/10.1038/s41598-019-47899-7

Holmes, N. D., Spatz, D. R., Oppel, S., Tershy, B., Croll, D. A., Keitt, B., Genovesi, P., Burfield, I. J., Will, D. J., Bond, A. L., Wegmann, A., Aguirre-Muñoz, A., Raine, A. F., Knapp, C. R., Hung, C.-H., Wingate, D., Hagen, E., Méndez-Sánchez, F., Rocamora, G., … Butchart, S. H. M. (2019). Globally important islands where eradicating invasive mammals will benefit highly threatened vertebrates. *PLoS ONE*, *14*(3), e0212128. https://doi.org/10.1371/journal.pone.0212128

Hone, J. (2007). *Analysis of Vertebrate Pest Control, Cambridge Studies in Applied Ecology and Resource Management*. https://www.booktopia.com.au/analysis-of-vertebrate-pest-control-jim-hone/book/9780521038973.html

Horsch, E. J., & Lewis, D. J. (2009). The Effects of Aquatic Invasive Species on Property Values: Evidence from a Quasi-Experiment. *Land Economics*, *85*(3), 391–409. https://doi.org/10.3368/le.85.3.391

Houngbo, S., Zannou, A., Aoudji, A., Sossou, H. C., Sinzogan, A., Sikirou, R., Zossou, E., Totin Vodounon, H. S., Adomou, A., & Ahanchédé, A. (2020). Farmers' knowledge and management practices of fall armyworm, *Spodoptera frugiperda* (J.E. Smith) in Benin, West Africa. *Agriculture (Switzerland)*, *10*(10), 1–15. https://doi.org/10.3390/agriculture10100430

Howald, G., Donlan, C. J., Galván, J. P., Russell, J. C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A., & Tershy, B. (2007). Invasive rodent eradication on islands. *Conservation Biology*, *21*(5), 1258–1268. https://doi.org/10.1111/j.1523-1739.2007.00755.x

Howard, P. L. (2019). Human adaptation to invasive species: A conceptual framework based on a case study metasynthesis. *Ambio*, *48*(12), 1401–1430. https://doi.org/10.1007/s13280-019-01297-5

Howarth, F. G. (1991). Environmental Impacts of Classical Biological Control. *Annual Review of Entomology*, *36*(1), 485–509. https://doi.org/10.1146/annurev.en.36.010191.002413

Hu, W., Bulusu, N., Chou, C. T., Jha, S., Taylor, A., & Tran, V. N. (2009). Design and evaluation of a hybrid sensor network for cane toad monitoring. *ACM Transactions on Sensor Networks*, *5*(1), 4:1-4:28. https://doi.org/10.1145/1464420.1464424

Huffaker, C. B., & Messenger, P. S. (Eds.). (1976). *Theory and Practice of Biological Control*. Elsevier. https://doi.org/10.1016/B978-0-12-360350-0.X5001-6

Hughes, K. A., Lee, J. E., Tsujimoto, M., Imura, S., Bergstrom, D. M., Ware, C., Lebouvier, M., Huiskes, A. H. L., Gremmen, N. J. M., Frenot, Y., Bridge, P. D., & Chown, S. L. (2011). Food for thought: Risks of non-native species transfer to the Antarctic region with fresh produce. *Biological Conservation*, *144*(5), 1682–1689. https://doi.org/10.1016/j.biocon.2011.03.001

Hulme, P. E. (2006). Beyond control: Wider implications for the management of biological invasions. *Journal of Applied Ecology*, *43*(5), 835–847. https://doi.org/10.1111/j.1365-2664.2006.01227.x

Hulme, P. E. (2009). Trade, transport and trouble: Managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, *46*(1), 10–18. https://doi.org/10.1111/j.1365-2664.2008.01600.x

Hulme, P. E. (2012). Weed risk assessment: A way forward or a waste of time? *Journal of Applied Ecology*, *49*(1), 10–19. https://doi.org/10.1111/j.1365-2664.2011.02069.x

Hulme, P. E. (2020a). One Biosecurity: A unified concept to integrate human, animal, plant, and environmental health. *Emerging Topics in Life Sciences*, *4*(5), 539–549. https://doi.org/10.1042/ETLS20200067

Hulme, P. E. (2020b). Plant invasions in New Zealand: Global lessons in prevention, eradication and control. *Biological Invasions*, *22*(5), 1539–1562. https://doi.org/10.1007/s10530-020-02224-6

Hulme, P. E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., Nentwig, W., Olenin, S., Panov, V., Pergl, J., Pyšek, P., Roques, A., Sol, D., Solarz, W., & Vilà, M. (2008). Grasping at the routes of biological invasions: A framework for integrating pathways into policy. *Journal of Applied Ecology*, *45*(2), 403–414. https://doi.org/10.1111/j.1365-2664.2007.01442.x

Hulme, P. E., Brundu, G., Carboni, M., Dehnen-Schmutz, K., Dullinger, S., Early, R., Essl, F., González-Moreno, P., Groom, Q. J., Kueffer, C., Kühn, I., Maurel, N., Novoa, A., Pergl, J., Pyšek, P., Seebens, H., Tanner, R., Touza, J. M., van Kleunen, M., & Verbrugge, L. N. H. (2018). Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions. *Journal of Applied Ecology*, *55*(1), 92–98. https://doi.org/10.1111/1365-2664.12953

Hulme, P. E., Pyšek, P., Nentwig, W., & Vilà, M. (2009). Will Threat of Biological Invasions Unite the European Union? *Science*, *324*(5923), 40–41. https://doi.org/10.1126/science.1171111

Hussain, M., Farooq, S., Merfield, C., & Jabran, K. (2018). Chapter 8—Mechanical Weed Control. In K. Jabran & B. S. Chauhan (Eds.), *Non-Chemical Weed Control* (pp. 133–155). Academic Press. https://doi.org/10.1016/B978-0-12-809881-3.00008-5

ICIPE. (2018). *Tackling invasive alien species in Africa*. http://www.icipe.org/news/tackling-invasive-alien-species-africa

IMO. (2004). *Adoption of the final act and any instruments, recommendations and resolutions resulting from the work of the conference*. International Conference on Ballast Water Management For Ships. https://www.liscr.com/sites/default/files/liscr_imo_resolutions/BWMConvention.pdf

Impson, F. A. C., Kleinjan, C. A., & Hoffmann, J. H. (2021). Suppression of seed production as a long-term strategy in weed biological control: The combined impact of two biocontrol agents on *Acacia mearnsii* in South Africa. *Biological Control*, *154*, 104503. https://doi.org/10.1016/j.biocontrol.2020.104503

Impson, F. A. C., Kleinjan, C. A., Hoffmann, J. H., Post, J. A., & Wood, A. R. (2011). Biological Control of Australian Acacia Species and *Paraserianthes lophantha* (Willd.) Nielsen (Mimosaceae) in South Africa. *African Entomology*, *19*(2), 186–207. https://doi.org/10.4001/003.019.0210

Ingold, T. (2000). Hunting and gathering as ways of perceiving the environment. In *The perception of the environment. Essays on livelihood, dwelling and skill*. Taylor & Francis e-Library, London. https://leiaarqueologia.files.wordpress.com/2017/08/the-perception-of-the-environment-tim-ingold.pdf

Inoue, M. N., Yokoyama, J., & Washitani, I. (2008). Displacement of Japanese native bumblebees by the recently introduced *Bombus terrestris* (L.) (Hymenoptera: Apidae). *Journal of Insect Conservation*, *12*(2), 135–146. https://doi.org/10.1007/s10841-007-9071-z

Ip, K. K. L., Liang, Y., Lin, L., Wu, H., Xue, J., & Qiu, J.-W. (2014). Biological control of invasive apple snails by two species of carp: Effects on non-target species matter. *Biological Control*, *71*, 16–22. https://doi.org/10.1016/j.biocontrol.2013.12.009

IPBES. (2016). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3235428

IPBES. (2018). *The IPBES assessment report on land degradation and restoration.* (p. 744). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237392

IPBES. (2019). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3831673

IPBES. (2020). *Report of the Indigenous and local knowledge dialogue workshop on the first order draft of the IPBES assessment of invasive alien species* (J. L. Andreve, R. Batzin Chojoj, A. Black, J. T. Cleofe, F. Daguitan, C. Grant, J. A. Guillao, L. Jacobs, T. Malcolm, L. Mulenkei, K. Kumar Rai, A. Nzovu, J. M. Ole Kaunga, M. E. Regpala, N. Sall, P. Shulbaeva, R. Spencer, P. Timoti, & Y. Upun, Eds.). https://ipbes.net/sites/default/files/inline-files/IPBES_IAS_2ndILKDialogue_FOD_Report_FINAL_ForWeb.pdf

IPBES. (2022a). *Methodological assessment of the diverse values and valuation of nature of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.6522522

IPBES. (2022b). *Report of the third Indigenous and local knowledge dialogue workshop for the IPBES thematic assessment of invasive alien species and their control.* https://ipbes.net/sites/default/files/2023-02/IPBES_IAS_3rdILKDialogue_SPM-SOD_Report_FinalForWeb2.pdf

IPCC. (2018). *Special Report Global warming of 1.5 °C*. https://www.ipcc.ch/sr15/

IPPC. (2017a). *Guidelines for the export, shipment, import and release of biological control agents and other beneficial organisms* (ISPM 3; International Standards for Phytosanitary Measures). International Plant Protection Convention. https://www.ippc.int/en/publications/guidelines-export-shipment-import-and-release-biological-control-agents-and-other/

IPPC. (2017b). *The use of integrated measures in a systems approach for pest risk management* (ISPM 14; International Standards for Phytosanitary Measures). International Plant Protection Convention. https://www.ippc.int/static/media/files/publication/en/2017/05/ISPM_14_2002_En_2017-05-25_PostCPM12_InkAm.pdf

IPPC. (2018). *Surveillance* (ISPM 6; International Standards for Phytosanitary Measures). FAO. https://www.ippc.int/static/media/files/publication/en/2018/06/ISPM_06_2018_En_Surveillance_2018-05-20_PostCPM13_KmRiysX.pdf

IPPC. (2019). *Framework for pest risk analysis* (ISPM 2; International Standards for Phytosanitary Measures). Secretariat of the International Plant Protection Convention (IPPC). https://www.fao.org/3/k0125e/k0125e.pdf

IPPC-CPM. (2020). *Reduce the incidence of contaminating pests associated with regulated articles and unregulated goods to protect plant health and facilitate trade (2019-002). Draft CPM recommendation.* International Plant Protection Convention. https://assets.ippc.int/static/media/files/publication/en/2021/01/16_CPM_2021_ANNEX2_2019-002_Draft_CPM_Rec_Contaminting_pests_2020-12-25.pdf

ISAC. (2017). *Advanced Biotechnology Tools for Invasive Species Management*. Invasive Species Advisory Committee. https://www.glc.org/wp-content/uploads/isac_advanced_biotechnology_white_paper.pdf

Island Conservation. (2018). *Data matters: Informing the eradication of invasive species on islands: North America and the Arctic region.* (Contractor's Report 2018-1.). National Invasive Species Council Secretariat. https://www.doi.gov/sites/doi.gov/files/uploads/data_matters_island_conservation_report.pdf

ISSG. (2018). Invasive alien species: The application of classical biological control for the management of established invasive alien species causing environmental impacts. In A. Sheppard, Q. Paynter, P. Mason, S. Murphy, P. Stoett, P. Cowan, J. Brodeur, K. Warner, C. Villegas, R. Shaw, H. Hinz, M. Hill, & P. Genovesi (Eds.), *Technical Report for the CBD. IUCN SSC Invasive Species Specialist Group.* (p. 96). https://www.cbd.int/doc/c/0c6f/7a35/eb8815eff54c3bc4a02139fd/cop-14-inf-09-en.pdf

IUCN. (2017). *Guidance for interpretation of CBD categories on introduction pathways* [Technical note prepared by IUCN for the European Commission]. International Union for Conservation of Nature. https://nora.nerc.ac.uk/id/eprint/519129/1/N519129CR.pdf

IUCN. (2018). *Compilation of costs of prevention and management of invasive alien species in the EU. Technical note prepared by IUCN for the European Commission.* https://circabc.europa.eu/sd/a/7b04a898-12e3-48c3-a0e5-f21a165259b4/2018-Compilation%20of%20costs%20of%20prevention%20and%20management%20of%20IAS%20in%20the%20EU.pdf

IUCN. (2020a). *IUCN EICAT Categories and Criteria. The Environmental Impact Classification for Alien Taxa.* (First edition). IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2020.05.en

IUCN. (2020b, June 9). *Management of IAS*. IUCN. https://www.iucn.org/regions/europe/our-work/biodiversity-conservation/invasive-alien-species/eu-regulation-technical-support/management-ias

IUCN/SSC. (2013). *Guidelines for Reintroductions and Other Conservation Translocations. Version 1.0.* IUCN Species Survival Commission. https://portals.iucn.org/library/sites/library/files/documents/2013-009.pdf

Januchowski-Hartley, S. R., Visconti, P., & Pressey, R. L. (2011). A systematic approach for prioritizing multiple management actions for invasive species. *Biological Invasions*, *13*(5), 1241–1253. https://doi.org/10.1007/s10530-011-9960-7

Jarić, I., Courchamp, F., Correia, R. A., Crowley, S. L., Essl, F., Fischer, A., González-Moreno, P., Kalinkat, G., Lambin, X., Lenzner, B., Meinard, Y., Mill, A., Musseau, C., Novoa, A., Pergl, J., Pyšek, P., Pyšková, K., Robertson, P., Schmalensee, M., … Jeschke, J. M. (2020). The role of species charisma in biological invasions. *Frontiers in Ecology and the Environment*, *18*(6), 345–353. https://doi.org/10.1002/fee.2195

Jarrad, F., Low-Choy, S., & Mengersen, K. (Eds.). (2015). *Biosecurity Surveillance: Quantitative Approaches*. CABI.

Jellinek, S., Rumpff, L., Driscoll, D. A., Parris, K. M., & Wintle, B. A. (2014). Modelling the benefits of habitat restoration in socio-ecological systems. *Biological Conservation*, *169*, 60–67. https://doi.org/10.1016/j.biocon.2013.10.023

Jia, S., & Gao, H. (2020). Review of Crop Disease and Pest Image Recognition Technology. *IOP Conference Series: Materials Science and Engineering*, *799*, 012045. https://doi.org/10.1088/1757-899X/799/1/012045

Johnson, B. A., Mader, A. D., Dasgupta, R., & Kumar, P. (2020). Citizen science and invasive alien species: An analysis of citizen science initiatives using information and communications technology (ICT) to collect invasive alien species observations. *Global Ecology and Conservation*, *21*, e00812. https://doi.org/10.1016/j.gecco.2019.e00812

Jones, B. A., McDermott, S. M., & Chermak, J. M. (2016). PLAN or get SLAM'ed: Optimal management of invasive species in the presence of indirect health externalities. *Journal of Environmental Management*, *180*, 538–550. https://doi.org/10.1016/j.jenvman.2016.05.026

Jones, C. M., Jones, S., Petrasova, A., Petras, V., Gaydos, D., Skrip, M. M., Takeuchi, Y., Bigsby, K., & Meentemeyer, R. K. (2021). Iteratively forecasting biological invasions with PoPS and a little help from our friends. *Frontiers in Ecology and the Environment*, *19*(7), 411–418. https://doi.org/10.1002/fee.2357

Jones, C., Moss, K., & Sanders, M. (2005). Diet of hedgehogs (*Erinaceus europaeus*) in the upper Waitaki Basin, New Zealand: Implications for conservation. *New Zealand Journal of Ecology*, *29*(1), 29–35. https://www.jstor.org/stable/24056190

Jones, H. P., Holmes, N. D., Butchart, S. H. M., Tershy, B. R., Kappes, P. J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D. P., Bonnaud, E., Burbidge, A. A., Campbell, K., Courchamp, F., Cowan, P. E., Cuthbert, R. J., Ebbert, S., Genovesi, P., Howald, G. R., Keitt, B. S., Kress, S. W., … Croll, D. A. (2016). Invasive mammal eradication on islands results in substantial conservation gains. *Proceedings of the National Academy of Sciences*, *113*(15), 4033–4038. https://doi.org/10.1073/pnas.1521179113

Joshi, C., De Leeuw, J., van Andel, J., Skidmore, A. K., Lekhak, H. D., van Duren, I. C., & Norbu, N. (2006). Indirect remote sensing of a cryptic forest understorey invasive species. *Forest Ecology and Management*, *225*(1), 245–256. https://doi.org/10.1016/j.foreco.2006.01.013

Joshi, C., de Leeuw, J., & van Duren, I. C. (2004). Remote sensing and GIS applications for mapping and spatial modelling of invasive species. *ISPRS 2004: Proceedings of the XXth ISPRS Congress*, *35*, 669–677. http://www.itc.nl/library/Papers_2004/peer_conf/joshi.pdf

Joshi, S., Bhaskar, H., Poon, V. A., Mala, B. J., Jayanthi, P. K., Pai, S. G., Thite, S. V., Sood, A. K., Kedar, S. C., Sridhar, V., Deepthy, K. B., Navik, O., & Rachana, R. R. (2021). Occurrence and spread of *Ceroplastes cirripediformis* Comstock (Hemiptera: Coccomorpha: Coccidae) in India. *Zootaxa*, *5039*(4), 561–570. https://doi.org/10.11646/zootaxa.5039.4.7

Jubase, N., Renteria, J. L., Maphisa, D., & Van Wyk, E. (2019). *Asphodelus fistulous* L., a newly discovered plant invader in South Africa: Assessing the risk of invasion and potential for eradication. *Bothalia*, *49*(1). https://doi.org/10.4102/abc.v49i1.2372

Julien, M. H., McFadyen, R. E., & Cullen, J. M. (Eds.). (2012). *Biological control of weeds in Australia*. CSIRO Publishing. http://public.eblib.com/choice/publicfullrecord.aspx?p=871257

Julien, M. H., & White, G. (1997). *Biological control of weeds: Theory and practical application* (49; ACIAR Monograph Series, p. 192). Australian Centre for International Agricultural Research. https://www.aciar.gov.au/publication/biological-control-weeds-theory-and-practical-application

Jurdak, R., Elfes, A., Kusy, B., Tews, A., Hu, W., Hernandez, E., Kottege, N., & Sikka, P. (2015). Autonomous surveillance for biosecurity. *Trends in Biotechnology*, *33*(4), 201–207. https://doi.org/10.1016/j.tibtech.2015.01.003

Jurdak, R., Sommer, P., Kusy, B., Kottege, N., Crossman, C., Mckeown, A., & Westcott, D. (2013). Camazotz: Multimodal activity-based GPS sampling. *Proceedings of the 12th International Conference on Information Processing in Sensor Networks*, 67–78. https://doi.org/10.1145/2461381.2461393

Kadoya, T., & Washitani, I. (2010). Predicting the rate of range expansion of an invasive alien bumblebee (*Bombus terrestris*) using a stochastic spatio-temporal model. *Biological Conservation*, *143*(5), 1228–1235. https://doi.org/10.1016/j.biocon.2010.02.030

Kah, M. (2015). Nanopesticides and Nanofertilizers: Emerging Contaminants or Opportunities for Risk Mitigation? *Frontiers in Chemistry*, *3*, 64. https://doi.org/10.3389/fchem.2015.00064

Kalaris, T., Fieselmann, D., Magarey, R., Colunga-Garcia, M., Roda, A., Hardie, D., Cogger, N., Hammond, N., Martin, P. A. T., & Whittle, P. (2014). The Role of Surveillance Methods and Technologies in Plant Biosecurity. In G. Gordh & S. McKirdy (Eds.), *The Handbook of Plant Biosecurity* (pp. 309–337). Springer Netherlands. https://doi.org/10.1007/978-94-007-7365-3_11

Kalumanga, E. (2015). *Why Integrated Natural Resource Management in Malagarasi-Muyovozi Ramsar Site, Tanzania? : Insights from Ugalla Game Reserve*. http://urn.kb.se/resolve?urn=urn:nbn:se:su:diva-116284

Kalwij, J. M., Steyn, C., & le Roux, P. C. (2014). Repeated monitoring as an effective early detection means: First records of naturalised *Solidago gigantea* Aiton (Asteraceae) in southern Africa. *South African Journal of Botany*, *93*, 204–206. https://doi.org/10.1016/j.sajb.2014.04.013

Kamigawara, K., Nakai, K., Noma, N., Hieda, S., Sarat, E., Dutartre, A., Renals, T., Bullock, R., Haury, J., Bottner, B., & Damien, J.-P. (2020). What kind of legislation can contribute to on-site management?: Comparative case studies on legislative developments in managing aquatic invasive alien plants in France, England, and Japan. *Journal of International Wildlife Law & Policy*, *23*(2), 83–108. https://doi.org/10.1080/13880292.2020.1788778

Kanavy, D., & Serr, M. (2017). *Sry* gene drive for rodent control: Reply to Gemmell and Tompkins. *Trends in Ecology & Evolution*, *32*(5), 315–316. https://doi.org/10.1016/j.tree.2017.03.006

Kannan, R., Shackleton, C. M., Krishnan, S., & Shaanker, R. U. (2016). Can local use assist in controlling invasive alien species in tropical forests? The case of *Lantana camara* in southern India. *Forest Ecology and Management*, *376*, 166–173. https://doi.org/10.1016/j.foreco.2016.06.016

Kansiime, M. K., Mugambi, I., Rwomushana, I., Nunda, W., Lamontagne-Godwin, J., Rware, H., Phiri, N. A., Chipabika, G., Ndlovu, M., & Day, R. (2019). Farmer perception of fall armyworm (*Spodoptera frugiderda* J.E. Smith) and farm-level management practices in Zambia. *Pest Management Science*, *75*(10), 2840–2850. https://doi.org/10.1002/ps.5504

Karl, B. J., & Best, H. A. (1982). Feral cats on Stewart Island; their foods, and their effects on kakapo. *New Zealand Journal of Zoology*, *9*(2), 287–293. https://doi.org/10.1080/03014223.1982.10423857

Kati, V., Hovardas, T., Dieterich, M., Ibisch, P. L., Mihok, B., & Selva, N. (2015). The challenge of implementing the European network of protected areas Natura 2000. *Conservation Biology*, *29*(1), 260–270. https://doi.org/10.1111/cobi.12366

Katsanevakis, S., Genovesi, P., Gaiji, S., Hvid, H. N., Roy, H. E., Nunes, A. L., Aguado, F. S., Bogucarskis, K., Debusscher, B., Deriu, I., Harrower, C., Josefsson, M., Lucy, F., Marchini, A., Richards, G., Trichkova, T., Vanderhoeven, S., Zenetos, A., & Cardoso, A. C. (2013). Implementing the European policies for alien species – networking, science, and partnership in a complex environment. *Management of Biological Invasions*, *4*(1), 3–6. https://doi.org/10.3391/mbi.2013.4.1.02

Kaufmann, J. C. (2004). Prickly Pear Cactus and Pastoralism in Southwest Madagascar. *Ethnology*, *43*(4), 345–361. https://doi.org/10.2307/3774032

Kay, B. H., & Russell, R. C. (Eds.). (2013). *Mosquito Eradication: The story of killing Campto*. CSIRO Publishing.

Keanly, C., & Robinson, T. B. (2020). Encapsulation as a biosecurity tool for managing fouling on recreational vessels. *Aquatic Invasions*, *15*(1), 81–97. https://doi.org/10.3391/ai.2020.15.1.06

Keeling, M. J., Franklin, D. N., Datta, S., Brown, M. A., & Budge, G. E. (2017). Predicting the spread of the Asian hornet (*Vespa velutina*) following its incursion into Great Britain. *Scientific Reports*, *7*(1), 6240. https://doi.org/10.1038/s41598-017-06212-0

Keith, I., Dawson, T. P., Collins, K. J., & Campbell, M. L. (2016). Marine invasive species: Establishing pathways, their presence and potential threats in the Galapagos Marine Reserve. *Pacific Conservation Biology*, *22*(4), 377–385. https://doi.org/10.1071/PC15020

Keith, J. M., & Spring, D. (2013). Agent-based Bayesian approach to monitoring the progress of invasive species eradication programs. *Proceedings of the National Academy of Sciences*, *110*(33), 13428–13433. https://doi.org/10.1073/pnas.1216146110

Keller, R. P., Frang, K., & Lodge, D. M. (2008). Preventing the Spread of Invasive Species: Economic Benefits of Intervention Guided by Ecological Predictions: *Economic Benefits of Invasion Prevention*. *Conservation Biology*, *22*(1), 80–88. https://doi.org/10.1111/j.1523-1739.2007.00811.x

Kellner, J. R., Asner, G. P., Kinney, K. M., Loarie, S. R., Knapp, D. E., Kennedy-Bowdoin, T., Questad, E. J., Cordell, S., & Thaxton, J. M. (2011). Remote analysis of biological invasion and the impact of enemy release. *Ecological Applications: A Publication of the Ecological Society of America*, *21*(6), 2094–2104. https://doi.org/10.1890/10-0859.1

Kelsch, A., Takahashi, Y., Dasgupta, R., Mader, A. D., Johnson, B. A., & Kumar, P. (2020). Invasive alien species and local communities in socio-ecological production landscapes and seascapes: A systematic review and analysis. *Environmental Science & Policy*, *112*, 275–281. https://doi.org/10.1016/j.envsci.2020.06.014

Kenis, M., Li, H., Fan, J., Courtial, B., Auger-Rozenberg, M.-A., Yart, A., Eschen, R., & Roques, A. (2018). Sentinel nurseries to assess the phytosanitary risks from insect pests on importations of live plants. *Scientific Reports*, *8*(1), 11217. https://doi.org/10.1038/s41598-018-29551-y

Kettenring, K. M., & Adams, C. R. (2011). Lessons learned from invasive plant control experiments: A systematic review and meta-analysis. *Journal of Applied Ecology*, *48*(4), 970–979. https://doi.org/10.1111/j.1365-2664.2011.01979.x

Kgori, P. M., Modo, S., & Torr, S. J. (2006). The use of aerial spraying to eliminate tsetse from the Okavango Delta of Botswana. *Acta Tropica*, *99*(2), 184–199. https://doi.org/10.1016/j.actatropica.2006.07.007

Khajuria, C., Ivashuta, S., Wiggins, E., Flagel, L., Moar, W., Pleau, M., Miller, K., Zhang, Y., Ramaseshadri, P., Jiang, C., Hodge, T., Jensen, P., Chen, M., Gowda, A., McNulty, B., Vazquez, C., Bolognesi, R., Haas, J., Head, G., & Clark, T. (2018). Development and characterization of the first dsRNA-resistant insect population from western corn rootworm, *Diabrotica virgifera virgifera* LeConte. *PLoS ONE*, *13*(5), e0197059. https://doi.org/10.1371/journal.pone.0197059

Kim, D. H., & Rossi, J. J. (2008). RNAi mechanisms and applications. *BioTechniques*, *44*(5), 613–616. https://doi.org/10.2144/000112792

Kim, J. H. K., Corson, P., Mulgan, N., & Russell, J. C. (2020). Rapid eradication assessment (REA): A tool for pest absence confirmation. *Wildlife Research*, *47*(2), 128–136. https://doi.org/10.1071/WR18154

King, C. M., & Powell, R. A. (2011). Managing an invasive predator pre-adapted to a pulsed resource: A model of stoat (*Mustela erminea*) irruptions in New Zealand beech forests. *Biological Invasions*, *13*(12), 3039–3055. https://doi.org/10.1007/s10530-011-9993-y

King, S., Drlik, T., Simon, L., & Quarles, W. (1996). Integrated weed management of gorse. *IPM Practitioner*, *18*(10), pp.1-9. https://www.cabdirect.org/cabdirect/abstract/19962302800

Kinney, G. F., & Wiruth, A. D. (1976). *Practical Risk Analysis for Safety Management*. Naval Weapons Center, China Lake, CA, https://www.researchgate.net/profile/Guibert-Crevecoeur/post/What-is-the-unification-theory-of-safety-science/attachment/59d6252c79197b80779837dc/AS%3A316678640996352%4014525 13479333/download/Basic_Kinney_article.pdf

Kirk, N., Kannemeyer, R., Greenaway, A., MacDonald, E., & Stronge, D. (2019). Understanding attitudes on new technologies to manage invasive species. *Pacific Conservation Biology*, *26*(1), 35–44. https://doi.org/10.1071/PC18080

Kittelson, P. M., & Boyd, M. J. (1997). Mechanisms of Expansion for an Introduced Species of Cordgrass, *Spartina densiflora*, in Humboldt Bay, California. *Estuaries*, *20*(4), 770–778. https://doi.org/10.2307/1352250

Kleitou, P., Rees, S., Cecconi, F., Kletou, D., Savva, I., Cai, L. L., & Hall-Spencer, J. M. (2021). Regular monitoring and targeted removals can control lionfish in Mediterranean Marine Protected Areas. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *31*(10), 2870–2882. https://doi.org/10.1002/aqc.3669

Kleitou, P., Savva, I., Kletou, D., Hall-Spencer, J. M., Antoniou, C., Christodoulides, Y., Chartosia, N., Hadjioannou, L., Dimitriou, A. C., Jimenez, C., Petrou, A., Sfenthourakis, S., & Rees, S. (2019). Invasive lionfish in the Mediterranean: Low public awareness yet high stakeholder concerns. *Marine Policy*, *104*, 66–74. https://doi.org/10.1016/j.marpol.2019.02.052

Knobloch, H., Köhler, H., Commander, N., Reinhold, P., Turner, C., & Chambers, M. (2009). Volatile Organic Compound (VOC) Analysis For Disease Detection: Proof Of Principle For Field Studies Detecting Paratuberculosis And Brucellosis. *AIP Conference Proceedings*, *1137*(1), 195–197. https://doi.org/10.1063/1.3156505

Koch, B. J., Hungate, B. A., & Price, L. B. (2017). Food-animal production and the spread of antibiotic resistance: The role of ecology. *Frontiers in Ecology and the Environment*, *15*(6), 309–318. https://doi.org/10.1002/fee.1505

Koehn, J. D., Todd, C. R., Zampatti, B. P., Stuart, I. G., Conallin, A., Thwaites, L., & Ye, Q. (2018). Using a Population Model to Inform the Management of River Flows and Invasive Carp (*Cyprinus carpio*). *Environmental Management*, *61*(3), 432–442. https://doi.org/10.1007/s00267-017-0855-y

Koffi, D., Kyerematen, R., Eziah, V. Y., Osei-Mensah, Y. O., Afreh-Nuamah, K., Aboagye, E., Osae, M., & Meagher, R. L. (2020). Assessment of impacts of fall armyworm, *Spodoptera frugiperda* (Lepidoptera: Noctuidae) on maize production in Ghana. *Journal of Integrated Pest Management*, *11*(1), 20. https://doi.org/10.1093/jipm/pmaa015

Koichi, K., Sangha, K. K., Cottrell, A., & Gordon, I. J. (2012). Aboriginal Rangers' Perspectives on Feral Pigs: Are they a Pest or a Resource? A Case Study in the Wet Tropics World Heritage Area of Northern Queensland. *Journal of Australian Indigenous Issues*, *15*(1), 2–20. https://www.researchgate.net/profile/Kamaljit-Sangha-2/publication/260752873_Aboriginal_Rangers'_Perspectives_on_Feral_Pigs_Are_they_a_Pest_or_a_Resource_A_Case_Study_in_the_Wet_Tropics_World_Heritage_Area_of_Northern_Queensland/links/00b7d53224cfdb852f000000/Aboriginal-Rangers-Perspectives-on-Feral-Pigs-Are-they-a-Pest-or-a-Resource-A-Case-Study-in-the-Wet-Tropics-World-Heritage-Area-of-Northern-Queensland.pdf

Koike, F., & Iwasaki, K. (2011). A simple range expansion model of multiple pathways: The case of nonindigenous green crab *Carcinus aestuarii* in Japanese waters. *Biological Invasions*, *13*(2), 459–470. https://doi.org/10.1007/s10530-010-9841-5

Kolar, C. S., & Lodge, D. M. (2002). Ecological Predictions and Risk Assessment for Alien Fishes in North America. *Science*, *298*(5596), 1233–1236. JSTOR. https://doi.org/10.1126/science.1075753

Kolpin, D. W., Thurman, E. M., & Linhart, S. M. (1998). The Environmental Occurrence of Herbicides: The Importance of Degradates in Ground Water. *Archives of Environmental Contamination and Toxicology*, *35*(3), 385–390. https://doi.org/10.1007/s002449900392

Kong, J., Cui, J., Wu, D., & Gerla, M. (2005). Building underwater ad-hoc networks and sensor networks for large scale real-time aquatic applications. *MILCOM 2005 – 2005 IEEE Military Communications Conference*, *3*, 1535–1541. https://doi.org/10.1109/MILCOM.2005.1605894

Kottege, N., Kroon, F., Jurdak, R., & Jones, D. (2012). Classification of Underwater Broadband Bio-acoustics Using Spectro-temporal Features. *Proceedings of the Seventh ACM International Conference on Underwater Networks and Systems*, 19:1-19:8. https://doi.org/10.1145/2398936.2398961

Kovacs, K. F., Mercader, R. J., Haight, R. G., Siegert, N. W., McCullough, D. G., & Liebhold, A. M. (2011). The influence of satellite populations of emerald ash borer on projected economic costs in U.S. communities, 2010–2020. *Journal of Environmental Management*, *92*(9), 2170–2181. https://doi.org/10.1016/j.jenvman.2011.03.043

Kriticos, D. J., Beautrais, J. R., & Dodd, M. B. (2018). WRASP: A spatial strategic weed risk analysis tool reveals important subnational variations in weed risks. *Weed Research*, *58*(6), 398–412. https://doi.org/10.1111/wre.12327

Kriticos, D. J., Brown, J., Radford, I., & Nicholas, M. (1999). Plant Population Ecology and Biological Control: *Acacia nilotica* as a Case Study. *Biological Control*, *16*(2), 230–239. https://doi.org/10.1006/bcon.1999.0746

Kriticos, D. J., De Barro, P. J., Yonow, T., Ota, N., & Sutherst, R. W. (2020). The potential geographical distribution and phenology of *Bemisia tabaci* Middle East/Asia Minor 1, considering irrigation and glasshouse production. *Bulletin of Entomological Research*, *110*(5), 567–576. https://doi.org/10.1017/S0007485320000061

Kriticos, D. J., Kean, J. M., Phillips, C. B., Senay, S. D., Acosta, H., & Haye, T. (2017). The potential global distribution of the brown marmorated stink bug, *Halyomorpha halys*, a critical threat to plant biosecurity. *Journal of Pest Science*, *90*(4), 1033–1043. https://doi.org/10.1007/s10340-017-0869-5

Kriticos, D. J., Morin, L., Leriche, A., Anderson, R. C., & Caley, P. (2013). Combining a Climatic Niche Model of an Invasive Fungus with Its Host Species Distributions to Identify Risks to Natural Assets: *Puccinia psidii* Sensu Lato in Australia. *PLoS ONE*, *8*(5), e64479. https://doi.org/10.1371/journal.pone.0064479

Kriticos, D. J., Sutherst, R. W., Brown, J. R., Adkins, S. W., & Maywald, G. F. (2003). Climate change and the potential distribution of an invasive alien plant: *Acacia nilotica* ssp. *indica* in Australia. *Journal of Applied Ecology*, *40*(1), 111–124. https://doi.org/10.1046/j.1365-2664.2003.00777.x

Kriticos, D. J., Watt, M. S., Withers, T. M., Leriche, A., & Watson, M. C. (2009). A process-based population dynamics model to explore target and non-target impacts of a biological control agent. *Ecological Modelling*, *220*(17), 2035–2050. https://doi.org/10.1016/j.ecolmodel.2009.04.039

Kriticos, D. J., Yonow, T., & McFadyen, R. E. (2005). The potential distribution of *Chromolaena odorata* (Siam weed) in relation to climate. *Weed Research*, *45*(4), 246–254. https://doi.org/10.1111/j.1365-3180.2005.00458.x

Krug, R. M., Roura-Pascual, N., & Richardson, D. M. (2009). Prioritising areas for the management of invasive alien plants in the CFR: Different strategies, different priorities? *South African Journal of Botany*, *75*(2), 408–409. https://doi.org/10.1016/j.sajb.2009.02.072

Kueffer, C., McDougall, K., Alexander, J., Daehler, C., Edwards, P., Haider, S., Milbau, A., Parks, C., Pauchard, A., Reshi, Z. A., Rew, L. J., Schroder, M., & Seipel, T. (2013). Plant Invasions into Mountain Protected Areas: Assessment, Prevention and Control at Multiple Spatial Scales. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7, pp. 89–113). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_6

Kumschick, S., Bacher, S., Bertolino, S., Blackburn, T. M., Evans, T., Roy, H. E., & Smith, K. (2020). Appropriate uses of EICAT protocol, data and classifications. *NeoBiota*, *62*, 193–212. https://doi.org/10.3897/neobiota.62.51574

Kumschick, S., Bacher, S., Dawson, W., Heikkilä, J., Sendek, A., Pluess, T., Robinson, T. B., & Kühn, I. (2012). A conceptual framework for prioritization of invasive alien species for management according to their impact. *NeoBiota*, *15*, 69–100. https://doi.org/10.3897/neobiota.15.3323

Kumschick, S., Foxcroft, L. C., & Wilson, J. R. (2020). Analysing the Risks Posed by Biological Invasions to South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 573–595). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_20

Kumschick, S., Wilson, J. R. U., & Foxcroft, L. C. (2020). A framework to support alien species regulation: The Risk Analysis for Alien Taxa (RAAT). *NeoBiota*, *62*, 213–239. https://doi.org/10.3897/neobiota.62.51031

Lacerda, A. C. R., Tomas, W. M., & Marinho-Filho, J. (2009). Domestic dogs as an edge effect in the Brasília National Park, Brazil: Interactions with native mammals. *Animal Conservation*, *12*(5), 477–487. https://doi.org/10.1111/j.1469-1795.2009.00277.x

Lafferty, K. D., & Kuris, A. M. (1996). Biological Control of Marine Pests. *Ecology*, *77*(7), 1989–2000. https://doi.org/10.2307/2265695

Lagos-Kutz, D., Voegtlin, D. J., Onstad, D., Hogg, D., Ragsdale, D., Tilmon, K., Hodgson, E., Difonzo, C., Groves, R., Krupke, C., Laforest, J., Seiter, N. J., Duerr, E., Bradford, B., & Hartman, G. L. (2020). The Soybean Aphid Suction Trap Network: Sampling the Aerobiological "Soup." *American Entomologist*, *66*(1), 48–55. https://doi.org/10.1093/ae/tmaa009

Lahdelma, R., Salminen, P., & Hokkanen, J. (2000). Using Multicriteria Methods in Environmental Planning and Management. *Environmental Management*, *26*(6), 595–605. https://doi.org/10.1007/s002670010118

Lampert, A., Hastings, A., Grosholz, E. D., Jardine, S. L., & Sanchirico, J. N. (2014). Optimal approaches for balancing invasive species eradication and endangered species management. *Science*, *344*(6187), 1028–1031. https://doi.org/10.1126/science.1250763

Lansink, A. O., Schut, M., Kamanda, J., & Klerkx, L. (2018). A multi-level and multi-actor approach to risk governance: A conceptual framework to support policy development for Ambrosia weed control. *Journal of Risk Research*, *21*(6), 780–799. https://doi.org/10.1080/13669877.2016.1247376

Lanzoni, A., Castoldi, A. F., Kass, G. E., Terron, A., De Seze, G., Bal-Price, A., Bois, F. Y., Delclos, K. B., Doerge, D. R., Fritsche, E., Halldorsson, T., Kolossa-Gehring, M., Hougaard Bennekou, S., Koning, F., Lampen, A., Leist, M., Mantus, E., Rousselle, C., Siegrist, M., … Younes, M. (2019). Advancing human health risk assessment. *EFSA Journal*, *17*(S1), e170712. https://doi.org/10.2903/j.efsa.2019.e170712

Laothawornkitkul, J., Moore, J. P., Taylor, J. E., Possell, M., Gibson, T. D., Hewitt, C. N., & Paul, N. D. (2008). Discrimination of Plant Volatile Signatures by an Electronic Nose: A Potential Technology for Plant Pest and Disease Monitoring. *Environmental Science & Technology*, *42*(22), 8433–8439. https://doi.org/10.1021/es801738s

Larson, D. L., Phillips-Mao, L., Quiram, G., Sharpe, L., Stark, R., Sugita, S., & Weiler, A. (2011). A framework for sustainable invasive species management: Environmental, social, and economic objectives. *Journal of Environmental Management*, *92*(1), 14–22. https://doi.org/10.1016/j.jenvman.2010.08.025

Larson, E. R., Graham, B. M., Achury, R., Coon, J. J., Daniels, M. K., Gambrell, D. K., Jonasen, K. L., King, G. D., LaRacuente, N., Perrin-Stowe, T. I., Reed, E. M., Rice, C. J., Ruzi, S. A., Thairu, M. W., Wilson, J. C., & Suarez, A. V. (2020). From eDNA to citizen science: Emerging tools for the early detection of invasive species. *Frontiers in Ecology and the Environment*, *18*(4), 194–202. https://doi.org/10.1002/fee.2162

Latinne, A., Hu, B., Olival, K. J., Zhu, G., Zhang, L., Li, H., Chmura, A. A., Field, H. E., Zambrana-Torrelio, C., Epstein, J. H., Li, B., Zhang, W., Wang, L.-F., Shi, Z.-L., & Daszak, P. (2020). Origin and cross-species transmission of bat coronaviruses in China. *Nature Communications*, *11*, 4235. https://doi.org/10.1038/s41467-020-17687-3

Latofski-Robles, M., Aguirre-Muñoz, A., Méndez-Sánchez, F., Reyes-Hernández, H., & Schlüter, S. (2014). Prioritizing Restoration Actions for the Islands of Mexico. *Monographs of the Western North American Naturalist*, *7*(1), 435–441. https://doi.org/10.3398/042.007.0133

Latofski-Robles, M., Méndez-Sánchez, F., Aguirre-Muñoz, A., Jáuregui-García, C., Koleff-Osorio, P., González-Martínez, A. I., Born-Schmidt, G., Bernal-Stoopen, J., & Rendón-Hernández, E. (2019). *Mexico's island biosecurity programme: Collaborative formulation and implementation*. 5. https://www.islas.org.mx/articulos_files/Latofski-Robles%202019.pdf

Latombe, G., Canavan, S., Hirsch, H., Hui, C., Kumschick, S., Nsikani, M. M., Potgieter, L. J., Robinson, T. B., Saul, W. -C., Turner, S. C., Wilson, J. R. U., Yannelli, F. A., & Richardson, D. M. (2019). A four-component classification of uncertainties in biological invasions: Implications for management. *Ecosphere*, *10*(4), e02669. https://doi.org/10.1002/ecs2.2669

Lau, H. Y., & Botella, J. R. (2017). Advanced DNA-Based Point-of-Care Diagnostic Methods for Plant Diseases Detection. *Frontiers in Plant Science*, *8*, 2016. https://doi.org/10.3389/fpls.2017.02016

Lawrence, A. (2006). 'No Personal Motive?' Volunteers, Biodiversity, and the False Dichotomies of Participation. *Ethics, Place & Environment*, *9*(3), 279–298. https://doi.org/10.1080/13668790600893319

Laycock, H., Moran, D., Smart, J., Raffaelli, D., & White, P. (2009). Evaluating the cost-effectiveness of conservation: The UK Biodiversity Action Plan. *Biological Conservation*, *142*(12), 3120–3127. https://doi.org/10.1016/j.biocon.2009.08.010

Lee, J. J. H., Frey, K., Fitch, R., & Sukkarieh, S. (2014, January 1). Fast path planning for precision weeding. *Proceedings of Australasian Conference on Robotics and Automation*. Australasian Conference on Robotics and Automation. https://opus.lib.uts.edu.au/handle/10453/106135

Legros, M., Marshall, J. M., Macfadyen, S., Hayes, K. R., Sheppard, A., & Barrett, L. G. (2021). Gene drive strategies of pest control in agricultural systems: Challenges and opportunities. *Evolutionary Applications*, *14*(9), 2162–2178. https://doi.org/10.1111/eva.13285

Lehmann, J. R. K., Prinz, T., Ziller, S. R., Thiele, J., Heringer, G., Meira-Neto, J. A. A., & Buttschardt, T. K. (2017). Open-Source Processing and Analysis of Aerial Imagery Acquired with a Low-Cost Unmanned Aerial System to Support Invasive Plant Management. *Frontiers in Environmental Science*, *5*. https://doi.org/10.3389/fenvs.2017.00044

Lehtiniemi, M., Ojaveer, H., David, M., Galil, B., Gollasch, S., McKenzie, C., Minchin, D., Occhipinti-Ambrogi, A., Olenin, S., & Pederson, J. (2015). Dose of truth-Monitoring marine non-indigenous species to serve legislative requirements. *Marine Policy*, *54*, 26–35. https://doi.org/10.1016/j.marpol.2014.12.015

Lenzner, B., Vicente, J., & Roy, H. E. (2021). *IPBES IAS assessment, data management report for Scenarios and Models Liaison*. https://doi.org/10.5281/ZENODO.5706521

Lessa, I., Guimarães, T. C. S., Bergallo, H. de G., Cunha, A., & Vieira, E. M. (2016). Domestic dogs in protected areas: A threat to Brazilian mammals? *Natureza & Conservação*, *14*(2), 46–56. https://doi.org/10.1016/j.ncon.2016.05.001

Leung, B., Roura-Pascual, N., Bacher, S., Heikkila, J., Brotons, L., Burgman, M. A., Dehnen-Schmutz, K., Essl, F., Hulme, P. E., Richardson, D. M., Sol, D., & Vilà, M. (2012). TEASIng apart alien species risk assessments: A framework for best practices. *Ecology Letters*, *15*(12), 1475–1493. https://doi.org/10.1111/ele.12003

Leung, B., Springborn, M. R., Turner, J. A., & Brockerhoff, E. G. (2014). Pathway-level risk analysis: The net present value of an invasive species policy in the US. *Frontiers in Ecology and the Environment*, *12*(5), 273–279. https://doi.org/10.1890/130311

Leung Brian, Lodge David M., Finnoff David, Shogren Jason F., Lewis Mark A., & Lamberti Gary. (2002). An ounce of prevention or a pound of cure: Bioeconomic risk analysis of invasive species. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, *269*(1508), 2407–2413. https://doi.org/10.1098/rspb.2002.2179

Lewis, J. S., Corn, J. L., Mayer, J. J., Jordan, T. R., Farnsworth, M. L., Burdett, C. L., VerCauteren, K. C., Sweeney, S. J., & Miller, R. S. (2019). Historical, current, and potential population size estimates of invasive wild pigs (*Sus scrofa*) in the United States. *Biological Invasions*, *21*(7), 2373–2384. https://doi.org/10.1007/s10530-019-01983-1

Li, K., Kusy, B., Jurdak, R., Ignjatovic, A.,

Kanhere, S. S., & Jha, S. (2014). K-FSOM: Fair Link Scheduling Optimization for Energy-Aware Data Collection in Mobile Sensor Networks. In B. Krishnamachari, A. L. Murphy, & N. Trigoni (Eds.), *Wireless Sensor Networks* (Vol. 8354, pp. 17–33). Springer International Publishing. https://doi.org/10.1007/978-3-319-04651-8_2

Li, Z., Zalucki, M. P., Yonow, T., Kriticos, D. J., Bao, H., Chen, H., Hu, Z., Feng, X., & Furlong, M. J. (2016). Population dynamics and management of diamondback moth (*Plutella xylostella*) in China: The relative contributions of climate, natural enemies and cropping patterns. *Bulletin of Entomological Research*, *106*(2), 197–214. https://doi.org/10.1017/S0007485315001017

Liebhold, A. M., Berec, L., Brockerhoff, E. G., Epanchin-Niell, R. S., Hastings, A., Herms, D. A., Kean, J. M., McCullough, D. G., Suckling, D. M., Tobin, P. C., & Yamanaka, T. (2016). Eradication of Invading Insect Populations: From Concepts to Applications. *Annual Review of Entomology*, *61*(1), 335–352. https://doi.org/10.1146/annurev-ento-010715-023809

Liebhold, A. M., & Kean, J. M. (2019). Eradication and containment of non-native forest insects: Successes and failures. *Journal of Pest Science*, *92*(1), 83–91. https://doi.org/10.1007/s10340-018-1056-z

Liebman, M., Mohler, C. L., & Staver, C. P. (2001). *Ecological Management of Agricultural Weeds*. Cambridge University Press.

LIFE STOPVESPA. (2021). *Vespa Velutina—The Project*. https://www.vespavelutina.eu/en-us/the-project/The-radar

Lim, C. H., Lee, J., Choi, Y., Park, J. W., & Kim, H. K. (2021). Advanced container inspection system based on dual-angle X-ray imaging method. *Journal of Instrumentation*, *16*(08), P08037. https://doi.org/10.1088/1748-0221/16/08/P08037

Lim, Z. X., Robinson, K. E., Jain, R. G., Sharath Chandra, G., Asokan, R., Asgari, S., & Mitter, N. (2016). Diet-delivered RNAi in *Helicoverpa armigera* – Progresses and challenges. *Journal of Insect Physiology*, *85*, 86–93. https://doi.org/10.1016/j.jinsphys.2015.11.005

Linke, L. M., Wilusz, J., Pabilonia, K. L., Fruehauf, J., Magnuson, R., Olea-Popelka, F., Triantis, J., Landolt, G., & Salman, M. (2016). Inhibiting avian influenza virus shedding using a novel RNAi antiviral vector technology: Proof of concept in an avian cell model. *AMB Express*, *6*(1), 16. https://doi.org/10.1186/s13568-016-0187-y

Liu, C., Wolter, C., Xian, W., & Jeschke, J. M. (2020). Species distribution models have limited spatial transferability for invasive species. *Ecology Letters*, *23*(11), 1682–1692. https://doi.org/10.1111/ele.13577

Liu, S., Maclean, K., & Robinson, C. (2019). A cost-effective framework to prioritise stakeholder participation options. *EURO Journal on Decision Processes*, *7*(3–4), 221–241. https://doi.org/10.1007/s40070-019-00103-7

Liu, S., Proctor, W., & Cook, D. (2010). Using an integrated fuzzy set and deliberative multi-criteria evaluation approach to facilitate decision-making in invasive species management. *Ecological Economics*, *69*(12), 2374–2382. https://doi.org/10.1016/j.ecolecon.2010.07.004

Liu, S., Sheppard, A., Kriticos, D. J., & Cook, D. (2011). Incorporating uncertainty and social values in managing invasive alien species: A deliberative multi-criteria evaluation approach. *Biological Invasions*, *13*(10), 2323–2337. https://doi.org/10.1007/s10530-011-0045-4

Liu, X., Blackburn, T. M., Song, T., Li, X., Huang, C., & Li, Y. (2019). Risks of biological invasion on the belt and road. *Current Biology*, *29*(3), 499-505.e4. https://doi.org/10.1016/j.cub.2018.12.036

Liu, X., Blackburn, T. M., Song, T., Wang, X., Huang, C., & Li, Y. (2020). Animal invaders threaten protected areas worldwide. *Nature Communications*, *11*(1), 2892. https://doi.org/10.1038/s41467-020-16719-2

Locke, A., Hanson, J. M., MacNair, N. G., & Smith, A. H. (2009). Rapid response to non-indigenous species. 2. Case studies of invasive tunicates in Prince Edward Island. *Aquatic Invasions*, *4*(1), 249–258. https://doi.org/10.3391/ai.2009.4.1.25

Lodge, D. M., Simonin, P. W., Burgiel, S. W., Keller, R. P., Bossenbroek, J. M., Jerde, C. L., Kramer, A. M., Rutherford, E. S., Barnes, M. A., Wittmann, M. E., Chadderton, W. L., Apriesnig, J. L., Beletsky, D., Cooke, R. M., Drake, J. M., Egan, S. P., Finnoff, D. C., Gantz, C. A., Grey, E. K., … Zhang, H. (2016). Risk Analysis and Bioeconomics of Invasive Species to Inform Policy and Management. *Annual Review of Environment and Resources*, *41*(1), 453–488. https://doi.org/10.1146/annurev-environ-110615-085532

Lohrmann, J., Cecchetto, N. R., Aizen, N., Arbetman, M. P., & Zattara, E. E. (2022). When bio is not green: The impacts of bumblebee translocation and invasion on native ecosystems. *CABI Reviews*, *2022*. https://doi.org/10.1079/cabireviews202217006

Longcore, T., Rich, C., & Sullivan, L. M. (2009). Critical Assessment of Claims Regarding Management of Feral Cats by Trap-Neuter-Return. *Conservation Biology*, *23*(4), 887–894. https://doi.org/10.1111/j.1523-1739.2009.01174.x

Lonsdale, W. M. (1994). Inviting trouble: Introduced pasture species in northern Australia. *Australian Journal of Ecology*, *19*(3), 345–354. https://doi.org/10.1111/j.1442-9993.1994.tb00498.x

Loope, L. L., Hughes, R. F., & Meyer, J.-Y. (2013). Plant Invasions in Protected Areas of Tropical Pacific Islands, with Special Reference to Hawaii. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas: Patterns, Problems and Challenges* (Vol. 7, pp. 313–348). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_15

Lopatin, J., Dolos, K., Kattenborn, T., & Fassnacht, F. E. (2019). How canopy shadow affects invasive plant species classification in high spatial resolution remote sensing. *Remote Sensing in Ecology and Conservation*, *5*(4), 302–317. https://doi.org/10.1002/rse2.109

Lopez, C. B., Cloern, J. E., Schraga, T. S., Little, A. J., Lucas, L. V., Thompson, J. K., & Burau, J. R. (2006). Ecological Values of Shallow-Water Habitats: Implications for the Restoration of Disturbed Ecosystems. *Ecosystems*, *9*(3), 422–440. https://doi.org/10.1007/s10021-005-0113-7

López, O., Rach, M. M., Migallon, H., Malumbres, M. P., Bonastre, A., & Serrano, J. J. (2012). Monitoring Pest Insect Traps by Means of Low-Power Image Sensor Technologies. *Sensors*, *12*(11), 15801–15819. https://doi.org/10.3390/s121115801

López-Gómez, M. J., Aguilar-Perera, A., & Perera-Chan, L. (2014). Mayan diver-fishers as citizen scientists: Detection and monitoring of the invasive red lionfish in the Parque Nacional Arrecife Alacranes, southern Gulf of Mexico. *Biological Invasions*, *16*(7), 1351–1357. https://doi.org/10.1007/s10530-013-0582-0

Louda, S. M., Pemberton, R. W., Johnson, M. T., & Follett, P. A. (2003). *Nontarget effects—The Achilles' heel of biological control? Retrospective analyses to reduce risk associated with biocontrol introductions* (Vol. 48). Annual Review of Entomology. https://doi.org/10.1146/annurev.ento.48.060402.102800

Lowe, S., Browne, M., Boudjelas, S., & De Poorter, M. (2000). *100 of the world's worst invasive alien species: A selection from the global invasive species database*. Invasive Species Specialist Group Auckland, New Zealand. www.issg.org/booklet.pdf

Lozier, J. D., Cameron, S. A., Duennes, M. A., Strange, J. P., Williams, P. H., Goulson, D., Brown, M. J. F., Morales, C., & Jepsen, S. (2015). Relocation risky for bumblebee colonies. *Science*, *350*(6258), 286–287. https://doi.org/10.1126/science.350.6258.286-b

Luaibi, A. R., Salman, T. M., & Miry, A. H. (2021). Detection of citrus leaf diseases using a deep learning technique. *International Journal of Electrical and Computer Engineering (IJECE)*, *11*(2), 1719–1727. https://doi.org/10.11591/ijece.v11i2.pp1719-1727

Lucy, F. E., Davis, E., Anderson, R., Booy, O., Bradley, K., Britton, J. R., Byrne, C., Caffrey, J. M., Coughlan, N. E., Crane, K., Cuthbert, R. N., Dick, J. T. A., Dickey, J. W. E., Fisher, J., Gallagher, C., Harrison, S., Jebb, M., Johnson, M., Lawton, C., … Trodd, W. (2020). Horizon scan of invasive alien species for the island of Ireland. *Management of Biological Invasions*, *11*(2), 155–177. https://doi.org/10.3391/mbi.2020.11.2.01

Lucy, F. E., Roy, H., Simpson, A., Carlton, J. T., Hanson, J. M., Magellan, K., Campbell, M. L., Costello, M. J., Pagad, S., Hewitt, C. L., McDonald, J., Cassey, P., Thomaz, S. M., Katsanevakis, S., Zenetos, A., Tricarico, E., Boggero, A., Groom, Q. J., Adriaens, T., … Panov, V. E. (2016). INVASIVESNET towards an international association for open knowledge on invasive alien species. *Management of Biological Invasions*, *7*(2), 131–139. https://doi.org/10.3391/mbi.2016.7.2.01

Lukey, P., & Hall, J. (2020). Biological Invasion Policy and Legislation Development and Implementation in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 515–551). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_18

Luna-Mendoza, L., Aguirre-Muñoz, A., Hernández-Montoya, J. C., Torres-Aguilar, M., García-Carreón, J. S., Puebla-Hernández, O., Luvianos-Colín, S., Cárdenas-Tapia, A., & Méndez-Sánchez, F. (2019). Ten years after feral goat eradication: The active restoration of plant communities on Guadalupe Island, Mexico. *Island Invasives: Scaling up to Meet the Challenge*, *62*, 571–575.

Lundgren, R. E., & McMakin, A. H. (2018). *Risk Communication A Handbook for Communicating Environmental, Safety, and Health Risks*. Wiley-IEEE Press. https://nbn-resolving.org/urn:nbn:de:101:1-2018080804355041270690

Lurgi, M., Wells, K., Kennedy, M., Campbell, S., & Fordham, D. A. (2016). A Landscape Approach to Invasive Species Management. *PLoS ONE*, *11*(7), e0160417. https://doi.org/10.1371/journal.pone.0160417

Lyon, A. (2010). *Review of Online Systems for Biosecurity Intelligence–Gathering and Analysis—ACERA Project 1003* (p. 99) [Final Report]. University of Maryland, College Park Australian National University University of Sydney. https://cebra.unimelb.edu.au/__data/assets/pdf_file/0005/2068691/1003-final-report.pdf

Lyons, J. E., Runge, M. C., Laskowski, H. P., & Kendall, W. L. (2008). Monitoring in the Context of Structured Decision-Making and Adaptive Management. *Journal of Wildlife Management*, *72*(8), 1683–1692. https://doi.org/10.2193/2008-141

Lyver, P. O., Akins, A., Phipps, H., Kahui, V., Towns, D. R., & Moller, H. (2016). Key biocultural values to guide restoration action and planning in New Zealand: Biocultural values for restoration. *Restoration Ecology*, *24*(3), 314–323. https://doi.org/10.1111/rec.12318

Mabin, C. A., Wilson, J. R. U., Le Roux, J. J., Majiedt, P., & Robinson, T. B. (2020). The first management of a marine invader in Africa: The importance of trials prior to setting long-term management goals. *Journal of Environmental Management*, *261*, 110213. https://doi.org/10.1016/j.jenvman.2020.110213

MacDonald, E. A., Balanovic, J., Edwards, E. D., Abrahamse, W., Frame, B., Greenaway, A., Kannemeyer, R., Kirk, N., Medvecky, F., Milfont, T. L., Russell, J. C., & Tompkins, D. M. (2020). Public Opinion Towards Gene Drive as a Pest Control Approach for Biodiversity Conservation and the Association of Underlying Worldviews. *Environmental Communication*, *14*(7), 904–918. https://doi.org/10.1080/17524032.2019.1702568

Machado, A. K., Brown, N. A., Urban, M., Kanyuka, K., & Hammond-Kosack, K. E. (2018). RNAi as an emerging approach to control Fusarium head blight disease and mycotoxin contamination in cereals. *Pest Management Science*, *74*(4), 790–799. https://doi.org/10.1002/ps.4748

Mack, R. N., & Lonsdale, W. M. (2001). Humans as global plant dispersers: Getting more than we bargained for: Current introductions of species for aesthetic purposes present the largest single challenge for predicting which plant immigrants will become future pests. *Bioscience*, *51*(2), 95–102. https://doi.org/10.1641/0006-3568(2001)051[0095:HAGPDG]2.0.CO;2

Mack, R. N., Simberloff, D., Lonsdale, W. M., Evans, H., Clout, M., & Bazzaz, F. A. (2000). Biotic invasions: Causes, epidemiology, global consequences, and control. *Ecological Applications*, *10*(3), 689–710. https://doi.org/10.1890/1051-0761(2000)010[0689:BICEGC]2.0.CO;2

Maclean, K., Farbotko, C., Mankad, A., Robinson, C. J., Curnock, M., Collins, K., & McAllister, R. R. J. (2018). Building social resilience to biological invasions. A case study of Panama Tropical Race 4 in the Australian Banana Industry. *Geoforum*, *97*, 95–105. https://doi.org/10.1016/j.geoforum.2018.10.018

Macnaughton, A. E., Carvajal-Vallejos, F. M., Argote, A., Rainville, T. K., Van Damme, P. A., & Carolsfeld, J. (2015). "*Paiche* reigns!" species introduction and indigenous fisheries in the Bolivian Amazon. *Maritime Studies*, *14*(1), 11. https://doi.org/10.1186/s40152-015-0030-0

Madegowda, C., & Rao, C. U. (2014). The Traditional Ecological Knowledge of Soliga Tribe on Eradication of *Lantana camara* and their Livelihood. *Antrocom Online Journal of Anthropology*, *10*(2), 163–173. http://www.antrocom.net/archives/2014/100214/05-Antrocom.pdf

Magona, N., Richardson, D. M., Le Roux, J. J., Kritzinger-Klopper, S., & Wilson, J. R. U. (2018). Even well-studied groups of alien species might be poorly inventoried: Australian *Acacia* species in South Africa as a case study. *NeoBiota*, *39*, 1–29. https://doi.org/10.3897/neobiota.39.23135

Mainali, K. P., Warren, D. L., Dhileepan, K., McConnachie, A., Strathie, L., Hassan, G., Karki, D., Shrestha, B. B., & Parmesan, C. (2015). Projecting future expansion of invasive species: Comparing and improving methodologies for species distribution modeling. *Global Change Biology*, *21*(12), 4464–4480. https://doi.org/10.1111/gcb.13038

Maistrello, L., Dioli, P., Bariselli, M., Mazzoli, G. L., & Giacalone-Forini, I. (2016). Citizen science and early detection of invasive species: Phenology of first occurrences of *Halyomorpha halys* in Southern Europe. *Biological Invasions*, *18*(11), 3109–3116. https://doi.org/10.1007/s10530-016-1217-z

Maldonado Andrade, G. (2019). *The Paradox of Culturally Useful Invasive Species: Chuspatel (Typha domingensis) Crafts of Lake Patzcuaro, Mexico* [M.A., California State University, Fullerton]. https://www.proquest.com/openview/57f112edbe28563f7b45504b68086835/1?pq-origsite=gscholar&cbl=18750&diss=y

Malmierca, L., Menvielle, M. F., Ramadori, D., Saavedra, B., Saunders, A., Soto, N., & Schiavini, A. (2011). Eradication of beaver (*Castor canadensis*), an ecosystem engineer and threat to southern Patagonia. *Island Invasives: Eradication and Management, International Conference on Island Invasives*, 87–90. https://www.researchgate.net/publication/288634952_Eradication_of_Beaver_Castor_canadensis_an_ecosystem_engineer_and_threat_to_southern_Patagonia

Maneetha, T. K., Sajeev, T. V., & Suganthasakthivel, R. (2017). Prediction modeling of Giant African Snail (*Achatina fulica*) infestation and evaluation of institutional mechanism for its management in Kerala. *Envis Newsletter*, *23*(1), 2–8. https://www.scribd.com/document/430630061/Envis-Paper-Maneetha

Manjunatha, S. B., Biradar, D. P., & Aladakatti, Y. R. (2016). Nanotechnology and its applications in agriculture: A review. *Journal of Farm Science*, *29*(1), 1–13. https://www.cabdirect.org/cabdirect/abstract/20163217790

Mankad, A., Hobman, E. V., & Carter, L. (2022). *Public perspectives towards using gene drive for invasive species management in Australia*. CSIRO. https://publications.csiro.au/rpr/ws/v1/download?pid=csiro:EP2022-1931&dsid=DS1

Mantelatto, M. C., Póvoa, A. A., Skinner, L. F., Araujo, F. V. de, & Creed, J. C. (2020). Marine litter and wood debris as habitat and vector for the range expansion of invasive corals (*Tubastraea* spp.). *Marine Pollution Bulletin*, *160*, 111659. https://doi.org/10.1016/j.marpolbul.2020.111659

Margerum, R. D., & Robinson, C. J. (Eds.). (2016). *The Challenges of Collaboration in Environmental Governance: Barriers and Responses*. Edward Elgar Publishing.

Marire, J. (2015). The Political Economy of South African Trout Fisheries. *Journal of Economic Issues*, *49*(1), 47–70. https://doi.org/10.1080/00213624.2015.1013878

Marshall, G. R., Coleman, M. J., Sindel, B. M., Reeve, I. J., & Berney, P. J. (2016). Collective action in invasive species control, and prospects for community-based governance: The case of serrated tussock (*Nassella trichotoma*) in New South Wales, Australia. *Land Use Policy*, *56*, 100–111. https://doi.org/10.1016/j.landusepol.2016.04.028

Marshall, N. A., Friedel, M., van Klinken, R. D., & Grice, A. C. (2011). Considering the social dimension of invasive species: The case of buffel grass. *Environmental Science & Policy*, *14*(3), 327–338. https://doi.org/10.1016/j.envsci.2010.10.005

Marten, A. L., & Moore, C. C. (2011). An options based bioeconomic model for biological and chemical control of invasive species. *Ecological Economics*, *70*(11), 2050–2061. https://doi.org/10.1016/j.ecolecon.2011.05.022

Martin, P. A. J. (2008). Current value of historical and ongoing surveillance for disease freedom: Surveillance for bovine Johne's disease in Western Australia. *Preventive Veterinary Medicine*, *84*(3–4), 291–309. https://doi.org/10.1016/j.prevetmed.2007.12.002

Martin, P. A. J., Cameron, A. R., & Greiner, M. (2007). Demonstrating freedom from disease using multiple complex data sources: 1: A new methodology based on scenario trees. *Preventive Veterinary Medicine*, *79*(2–4), 71–97. https://doi.org/10.1016/j.prevetmed.2006.09.008

Martin, P. A., Shackelford, G. E., Bullock, J. M., Gallardo, B., Aldridge, D. C., & Sutherland, W. J. (2020). Management of UK priority invasive alien plants: A systematic review protocol. *Environmental Evidence*, *9*(1), 1. https://doi.org/10.1186/s13750-020-0186-y

Martins, T. L. F., Brooke, M. de L., Hilton, G. M., Farnsworth, S., Gould, J., & Pain, D. J. (2006). Costing eradications of alien mammals from islands. *Animal Conservation*, *9*(4), 439–444. https://doi.org/10.1111/j.1469-1795.2006.00058.x

Marturana, F., Tacconi, S., & Italiano, G. F. (2015). A machine learning-based approach to digital triage. In A. T. S. Ho & S. Li (Eds.), *Handbook of Digital Forensics of Multimedia Data and Devices* (pp. 94–132). John Wiley & Sons, Ltd Chichester, UK. https://doi.org/10.1002/9781118705773.ch3

Mason, P. G. (Ed.). (2021). *Biological Control: Global Impacts, Challenges and Future Directions of Pest Management*. Csiro Publishing.

Mason, P. G., Kllapwijk, J. N., & Smith, D. (2021). Access and benefit-sharing and biological control. In P. G. Mason (Ed.), *Biological Control: Global impacts, Challenges and Future* (pp. 258–288). CSIRO Publishing. https://www.routledge.com/Biological-Control-Global-Impacts-Challenges-and-Future-Directions-of/Mason/p/book/9781032109275

Mason, T. J., Lonsdale, W. M., & French, K. (2005). Environmental weed control policy in Australia: Current approaches, policy limitations and future directions. *Pacific Conservation Biology*, *11*(4), 233–245. https://doi.org/10.1071/pc050233

Masuda, B. M., Fisher, P., & Beaven, B. (2015). Residue profiles of brodifacoum in coastal marine species following an island rodent eradication. *Ecotoxicology and Environmental Safety*, *113*, 1–8. https://doi.org/10.1016/j.ecoenv.2014.11.013

Matthies, M., Giupponi, C., & Ostendorf, B. (2007). Environmental decision support systems: Current issues, methods and tools. *Environmental Modelling & Software*, *22*(2), 123–127. https://doi.org/10.1016/j.envsoft.2005.09.005

Matzek, V., Covino, J., Funk, J. L., & Saunders, M. (2014). Closing the knowing-doing gap in invasive plant management: Accessibility and interdisciplinarity of scientific research. *Conservation Letters*, *7*(3), 208–215. https://doi.org/10.1111/conl.12042

Mayer, D. G., Atzeni, M. G., Stuart, M. A., Anaman, K. A., & Butler, D. G. (1998). Mating competitiveness of irradiated flies for screwworm fly eradication campaigns. *Preventive Veterinary Medicine*, *36*(1), 1–9. https://doi.org/10.1016/S0167-5877(98)00078-6

Mbaabu, P. R., Ng, W.-T., Schaffner, U., Gichaba, M., Olago, D., Choge, S., Oriaso, S., & Eckert, S. (2019). Spatial evolution of prosopis invasion and its effects on LULC and livelihoods in Baringo, Kenya. *Remote Sensing*, *11*(10), 1217. https://doi.org/10.3390/rs11101217

McAllister, R. R. J., Robinson, C. J., Brown, A., Maclean, K., Perry, S., & Liu, S. (2017). Balancing collaboration with coordination: Contesting eradication in the Australian plant pest and disease biosecurity system. *International Journal of the Commons*, *11*(1), Article 1. https://doi.org/10.18352/ijc.701

McAllister, R. R. J., Robinson, C. J., Maclean, K., Guerrero, A. M., Collins, K., Taylor, B. M., & De Barro, P. J. (2015). From local to central: A network analysis of who manages plant pest and disease outbreaks across scales. *Ecology and Society*, *20*(1), 67. https://doi.org/10.5751/ES-07469-200167

McCarthy, R. J., Levine, S. H., & Reed, J. M. (2013). Estimation of effectiveness of three methods of feral cat population control by use of a simulation model. *Journal of the American Veterinary Medical Association*, *243*(4), 502–511. https://doi.org/10.2460/javma.243.4.502

McCluskey, B. J., & Salman, M. D. (2003). Use of sentinel herds in monitoring and surveillance systems. In *Animal disease surveillance and survey systems: Methods and applications* (pp. 119–133). Wiley Online Library. https://doi.org/10.1002/9780470344866.ch8

McColl, K. A., Sunarto, A., & Holmes, E. C. (2016). Cyprinid herpesvirus 3 and its evolutionary future as a biological control agent for carp in Australia. *Virology Journal*, *13*(1), 206. https://doi.org/10.1186/s12985-016-0666-4

McFadyen, R. E. C. (1998). Biological Control of Weeds. *Annual Review of Entomology*, *43*(1), 369–393. https://doi.org/10.1146/annurev.ento.43.1.369

McGeoch, M. A., Butchart, S. H. M., Spear, D., Marais, E., Kleynhans, E. J., Symes, A., Chanson, J., & Hoffmann, M. (2010). Global indicators of biological invasion: Species numbers, biodiversity impact and policy responses. *Diversity and Distributions*, *16*(1), 95–108. https://doi.org/10.1111/j.1472-4642.2009.00633.x

McGeoch, M. A., Genovesi, P., Bellingham, P. J., Costello, M. J., McGrannachan, C., & Sheppard, A. (2016). Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion. *Biological Invasions*, *18*(2), 299–314. https://doi.org/10.1007/s10530-015-1013-1

McGeoch, M. A., Groom, Q. J., Pagad, S., Petrosyan, V., Ruiz, G., & Wilson, J. (2016). *Data fitness for use in research on alien and invasive species*. GBIF Secretariat, Copenhagen. http://www.gbif.org/document/82958

McGeoch, M. A., Spear, D., Kleynhans, E. J., & Marais, E. (2012). Uncertainty in invasive alien species listing. *Ecological Applications*, *22*(3), 959–971. https://doi.org/10.1890/11-1252.1

McGinn, D., & Disbury, M. (2019). National saltmarsh Mosquito Surveillance programme 2018-2019. *Surveillance*, *46*(3), 87–88.

McGlynn, T. P. (1999). The worldwide transfer of ants: Geographical distribution and ecological invasions. *Journal of Biogeography*, *26*(3), 535–548. https://doi.org/10.1046/j.1365-2699.1999.00310.x

McGrannachan, C. M., Pagad, S., & McGeoch, M. A. (2021). A multiregional assessment of transnational pathways of introduction. *NeoBiota*, *64*, 43–67. https://doi.org/10.3897/neobiota.64.60642

McInerney, J., Small, K., & Caley, P. (1995). Prevalence of *Mycobacterium bovis* infection in feral pigs in the Northern Territory. *Australian Veterinary Journal*, *72*(12), 448–451. https://doi.org/10.1111/j.1751-0813.1995.tb03486.x

McKenzie, C. H., Reid, V., Lambert, G., Matheson, K., Minchin, D., Pederson, J., Brown, L., Curd, A., Gollasch, S., Goulletquer, P., Occhipinti-Ambrogi, A., Simard, N., & Therriault, T. W. (2017). *Alien Species Alert: Didemnum vexillum Kott, 2002: Invasion, impact, and control* (ICES Cooperative Research Report No. 335). International Council for the Exploration of the Sea. http://doi.org/10.17895/ices.pub.2138

McLean, P., Gallien, L., Wilson, J. R. U., Gaertner, M., & Richardson, D. M. (2017). Small urban centres as launching sites for plant invasions in natural areas: Insights from South Africa. *Biological Invasions*, *19*(12), 3541–3555. https://doi.org/10.1007/s10530-017-1600-4

McLeod, L., & Saunders, G. (2013). *Pesticides used in the management of vertebrate pests in Australia: A review*. NSW Department of Primary industries.

McNie, E. C. (2007). Reconciling the supply of scientific information with user demands: An analysis of the problem and review of the literature. *Environmental Science & Policy*, *10*(1), 17–38. https://doi.org/10.1016/j.envsci.2006.10.004

Meat & Livestock Australia. (2021). *Australia still world's top goatmeat exporter | Meat & Livestock Australia*. MLA Corporate. https://www.mla.com.au/prices-markets/market-news/2021/australia-remains-top-goatmeat-exporter-despite-decline/

Meek, P. D., Ballard, G.-A., & Fleming, P. J. S. (2015). The pitfalls of wildlife camera trapping as a survey tool in Australia. *Australian Mammalogy*, *37*(1), 13–22. https://doi.org/10.1071/AM14023

Meier, S., Taff, G. N., Aune, J. B., & Eiter, S. (2017). Regulation of the Invasive Plant *Heracleum persicum* by Private Landowners in Tromsø, Norway. *Invasive Plant Science and Management*, *10*(2), 166–179. https://doi.org/10.1017/inp.2017.11

Merwe, J. V. (2015, November 9). *Barrow Island Quarantine: Beyond Best Practice*. Abu Dhabi International Petroleum Exhibition and Conference. https://doi.org/10.2118/177952-MS

Mesnage, R., & Antoniou, M. N. (2018). Ignoring Adjuvant Toxicity Falsifies the Safety Profile of Commercial Pesticides. *Frontiers in Public Health*, *5*, 361. https://doi.org/10.3389/fpubh.2017.00361

Meyer, J.-Y. (2008). Is eradication of the invasive tree miconia feasible? Lessons from 15 years of active management in French Polynesia (Pacific Islands). In R. D. Van Klinken, V. A. Osten, F. D. Panetta, & J. Scanlan (Eds.), *Proceedings of the 16th Australian Weeds Conference* (p. 433). https://caws.org.nz/old-site/awc/2008/awc200814331.pdf

Meyerson, L. A., & Reaser, J. K. (2002). Biosecurity: Moving toward a comprehensive approach: A comprehensive approach to biosecurity is necessary to minimize the risk of harm caused by non-native organisms to agriculture, the economy, the environment, and human health. *BioScience*, *52*(7), 593–600. https://doi.org/10.1641/0006-3568(2002)052[0593:BMTACA]2.0.CO;2

Middleton, K. (1999). Who Killed "Malagasy Cactus"? Science, Environment and Colonialism in Southern Madagascar (1924-1930). *Journal of Southern African Studies*, *25*(2), 215–248. http://www.jstor.org/stable/2637601

Migiro, L., & Otieno, W. (2020). The Role of Plantwise in Improving Detection and Action on Pest Situations. In S. Niassy, S. Ekesi, L. Migiro, & W. Otieno (Eds.), *Sustainable Management of Invasive Pests in Africa* (Vol. 14, pp. 245–251). Springer International Publishing. https://doi.org/10.1007/978-3-030-41083-4_19

Mill, A. C., Crowley, S. L., Lambin, X., McKinney, C., Maggs, G., Robertson, P., Robinson, N. J., Ward, A. I., & Marzano, M. (2020). The challenges of long-term invasive mammal management: Lessons from the UK. *Mammal Review*, *50*(2), 136–146. https://doi.org/10.1111/mam.12186

Miller, B. W., Breckheimer, I., McCleary, A. L., Guzmán-Ramirez, L., Caplow, S. C., Jones-Smith, J. C., & Walsh, S. J. (2010). Using stylized agent-based models for population–environment research: A case study from the Galápagos Islands. *Population and Environment*, *31*(6), 401–426. https://doi.org/10.1007/s11111-010-0110-4

Mills, N. J., & Getz, W. M. (1996). Modelling the biological control of insect pests: A review of host-parasitoid models. *Ecological Modelling*, *92*(2), 121–143. https://doi.org/10.1016/0304-3800(95)00177-8

Ministry for Primary Industries, New Zealand. (2020, November 16). *"Surveillance" biosecurity magazine | MPI – Ministry for Primary Industries. A New Zealand Government Department.* Ministry for Primary Industries. https://www.mpi.govt.nz/biosecurity/about-biosecurity-in-new-zealand/surveillance-biosecurity-magazine/

Ministry for Primary Industries, New Zealand. (2021). *Brown marmorated stink bug: Requirements for importers | MPI – Ministry for Primary Industries. A New Zealand Government Department.* Ministry for Primary Industries. https://www.mpi.govt.nz/import/vehicles-machinery-parts/brown-marmorated-stink-bug-requirements-for-importers/

Miththapala, S. (2007). *A strategy for addressing issues of aquatic invasive alien species in the Lower Mekong Basin.* (p. 14). IUCN: Mekong Wetland Biodiversity Programme and Regional Species Conservation Programme, The World Conservation Union (IUCN), Asia. https://portals.iucn.org/library/sites/library/files/documents/2007-074.pdf

Mitter, N., Worrall, E. A., Robinson, K. E., Xu, Z. P., & Carroll, B. J. (2017). Induction of virus resistance by exogenous application of double-stranded RNA. *Current Opinion in Virology*, *26*, 49–55. https://doi.org/10.1016/j.coviro.2017.07.009

Mo, J., Frank, E., & Vetrova, V. (2017). Large-Scale Automatic Species Identification. In W. Peng, D. Alahakoon, & X. Li (Eds.), *AI 2017: Advances in Artificial Intelligence* (pp. 301–312). Springer International Publishing. https://doi.org/10.1007/978-3-319-63004-5_24

Molina, V., & Drake, L. A. (2016). Efficacy of open-ocean ballast water exchange: A review. *Management of Biological Invasions*, *7*(4), 375–388. https://doi.org/10.3391/mbi.2016.7.4.07

Monterroso, I., Binimelis, R., & Rodríguez-Labajos, B. (2011). New methods for the analysis of invasion processes: Multi-criteria evaluation of the invasion of *Hydrilla verticillata* in Guatemala. *Journal of Environmental Management*, *92*(3), 494–507. https://doi.org/10.1016/j.jenvman.2010.09.017

Moon, K., Blackman, D. A., & Brewer, T. D. (2015). Understanding and integrating knowledge to improve invasive species management. *Biological Invasions*, *17*(9), 2675–2689. https://doi.org/10.1007/s10530-015-0904-5

Moon, K., Blackman, D., Brewer, T. D., & Sarre, S. D. (2017). Environmental governance for urgent and uncertain problems. *Biological Invasions*, *19*(3), 785–797. https://doi.org/10.1007/s10530-016-1351-7

Moran, V. C., Hoffmann, J. H., & Zimmermann, H. G. (2005). Biological control of invasive alien plants in South Africa: Necessity, circumspection, and success. *Frontiers in Ecology and the Environment*, *3*(2), 71–77. https://doi.org/10.1890/1540-9295(2005)003[0071:BCOIAP]2.0.CO;2

Moran, V. C., Hoffmann, J. H., & Zimmermann, H. G. (2013). 100 years of biological control of invasive alien plants in South Africa: History, practice and achievements : news & views. *South African Journal of Science*, *109*(9), a0022. https://doi.org/10.10520/EJC142072

More, S. J., Radunz, B., & Glanville, R. J. (2015). Lessons learned during the successful eradication of bovine tuberculosis from Australia. *Veterinary Record*, *177*(9), 224–232. https://doi.org/10.1136/vr.103163

Morin, L. (2020). Progress in Biological Control of Weeds with Plant Pathogens. *Annual Review of Phytopathology*, *58*(1), 201–223. https://doi.org/10.1146/annurev-phyto-010820-012823

Morin, L., Evans, K. J., & Sheppard, A. W. (2006). Selection of pathogen agents in weed biological control: Critical issues and peculiarities in relation to arthropod agents. *Australian Journal of Entomology*, *45*(4), 349–365. https://doi.org/10.1111/j.1440-6055.2006.00562.x

Morin, L., Reid, A. M., Sims-Chilton, N. M., Buckley, Y. M., Dhileepan, K., Hastwell, G. T., Nordblom, T. L., & Raghu, S. (2009). Review of approaches to evaluate the effectiveness of weed biological control agents. *Biological Control*, *51*(1), 1–15. https://doi.org/10.1016/j.biocontrol.2009.05.017

Moro, D., Byrne, M., Kennedy, M., Campbell, S., & Tizard, M. (2018). Identifying knowledge gaps for gene drive research to control invasive animal species: The next CRISPR step. *Global Ecology and Conservation*, *13*, e00363. https://doi.org/10.1016/j.gecco.2017.e00363

Morris, I. (1996). *Kakadu National Park, Australia.* Steve Parish Publishing.

Morrisey, D. J., Depree, C. V., Hickey, C. W., McKenzie, D. S., Middleton, I., Smith, M. D., Stewart, M., & Thompson, K. J. (2016). Rapid treatment of vessels fouled with an invasive polychaete, *Sabella spallanzanii*, using a floating dock and chlorine as a biocide. *Biofouling*, *32*(2), 135–144. https://doi.org/10.1080/08927014.2015.1126713

Morrisey, D. J., & Woods, C. (2015). *In-water cleaning technologies: Review of information*. Ministry for Primary Industries, Manatū Ahu Matua.

Moser, A. Y., Brown, W. Y., Bizo, L. A., Andrew, N. R., & Taylor, M. K. (2020). Biosecurity Dogs Detect Live Insects after Training with Odor-Proxy Training Aids: Scent Extract and Dead Specimens. *Chemical Senses*, *45*(3), 179–186. https://doi.org/10.1093/chemse/bjaa001

Moser, D., Lenzner, B., Weigelt, P., Dawson, W., Kreft, H., Pergl, J., Pyšek, P., van Kleunen, M., Winter, M., Capinha, C., Cassey, P., Dullinger, S., Economo, E. P., García-Díaz, P., Guénard, B., Hofhansl, F., Mang, T., Seebens, H., & Essl, F. (2018). Remoteness promotes biological invasions on islands worldwide. *Proceedings of the National Academy of Sciences*, *115*(37), 9270–9275. https://doi.org/10.1073/pnas.1804179115

Mosquito Alert. (2021). *Citizen science to investigate and control disease-carrying mosquitoes*. Mosquito Alert. http://www.mosquitoalert.com/en/

Mphande, F. A. (2016). Surveillance. In F. A. Mphande, *Infectious Diseases and Rural Livelihood in Developing Countries* (pp. 115–127). Springer Singapore. https://doi.org/10.1007/978-981-10-0428-5_7

Müllerová, J. (2019). UAS for Nature Conservation–Monitoring Invasive Species. In *Applications of Small Unmanned Aircraft Systems: Best Practices and Case Studies* (p. 290). CRC Press. http://dx.doi.org/10.1201/9780429244117-8

Müllerová, J., Bartaloš, T., Brůna, J., Dvořák, P., & Vítková, M. (2017). Unmanned aircraft in nature conservation: An example from plant invasions. *International Journal of Remote Sensing*, *38*(8–10), 2177–2198. https://doi.org/10.1080/01431161.2016.1275059

Mungi, N. A., Kaushik, M., Mohanty, N. P., Rastogi, R., Johnson, J. A., & Qureshi, Q. (2019). Identifying knowledge gaps in the research and management of invasive species in India. *Biologia*, *74*(6), 623–629. https://doi.org/10.2478/s11756-018-00186-8

Muraleedharan, P. K., & Anitha, V. (2000). The economic impact of *Mikania micrantha* on teak plantations in Kerala. *Indian Journal of Forestry*, *23*(3), 248–251.

Murdoch, W. W., Briggs, C. J., & Nisbet, R. M. (2003). Consumer-Resource Dynamics (MPB-36). In *Consumer-Resource Dynamics (MPB-36)*. Princeton University Press. https://doi.org/10.1515/9781400847259

Murdoch, W. W., Swarbrick, S. L., & Briggs, C. J. (2006). Biological control: Lessons from a study of California red scale. *Population Ecology*, *48*(4), 297–305. https://doi.org/10.1007/s10144-006-0004-6

Murphy, D., Gormley, E., Collins, D. M., McGrath, G., Sovsic, E., Costello, E., & Corner, L. A. L. (2011). Tuberculosis in cattle herds are sentinels for *Mycobacterium bovis* infection in European badgers (*Meles meles*): The Irish Greenfield Study. *Veterinary Microbiology*, *151*(1–2), 120–125. https://doi.org/10.1016/j.vetmic.2011.02.034

Murray, K., Jepson, P. C., & Chaola, M. (2019). *Fall Armyworm Management for Maize Smallholders in Malawi: An Integrated Pest Management Strategic Plan*. CDMX, CIMMYT. https://repository.cimmyt.org/handle/10883/20170

Murray, K., Jepson, P. C., & Huesing, J. (2021). *Fall armyworm for maize smallholders in Kenya: An integrated pest management strategic plan*. USAID, CIMMYT. https://repository.cimmyt.org/handle/10883/21259

Mutze, G. J. (1991). Long-term effects of warren ripping for rabbit control in semi-arid South Australia. *The Rangeland Journal*, *13*(2), 96–106. https://doi.org/10.1071/rj9910096

Mwangi, E., & Swallow, B. (2005). *Invasion of* Prosopis juliflora *and local livelihoods: Case study from the lake Baringo area of Kenya*. ICRAF Working Paper – no. 3. Nairobi: World Agroforestry Centre. http://apps.worldagroforestry.org/downloads/Publications/PDFS/WP13657.pdf

Myers, J. H., Savoie, A., & van Randen, E. (1998). Eradication and pest management. *Annual Review of Entomology*, *43*, 471–491. https://doi.org/10.1146/annurev.ento.43.1.471

Naidoo, R., & Ricketts, T. H. (2006). Mapping the Economic Costs and Benefits of Conservation. *PLoS Biology*, *4*(11), e360. https://doi.org/10.1371/journal.pbio.0040360

National Academies of Sciences, Engineering, and Medicine. (2016). *Gene Drives on the Horizon: Advancing Science, Navigating Uncertainty, and Aligning Research with Public Values*. National Academies Press. https://doi.org/10.17226/23405

Navarro, H., & Navarro, S. (2016). Current global challenges to the use of fumigants. In S. Navarro, D. S. Jayas, & K. Alagusundaram (Eds.), *Proceedings of the 10th International Conference on controlled atmosphere and fumigation in stored products* (pp. 126–133). CAF Permanent Committee Secretariat, CAF. http://ftic.co.il/2016NewDelhiPDF/PP126-133-A022-SESSION03.pdf

Nayak, M. K., Collins, P. J., & Pavic, H. (2003). Developments in phosphine resistance in China and possible implications for Australia. In E. J. Wright, M. C. Webb, & E. Highley (Eds.), *Stored grain in Australia 2003.* (pp. 156–159). CSIRO Stored Grain Research Laboratory. https://storedgrain.com.au/wp-content/uploads/2013/06/30.pdf

Newsome, T., van Eeden, L., Lazenby, B., & Dickman, C. (2017). Does Culling Work? *Australasian Science*, *38*(1), 28–30. http://www.australasianscience.com.au/article/issue-janfeb-2017/does-culling-work.html

Niemiec, R. M., Pech, R. P., Norbury, G. L., & Byrom, A. E. (2017). Landowners' Perspectives on Coordinated, Landscape-Level Invasive Species Control: The Role of Social and Ecological Context. *Environmental Management*, *59*(3), 477–489. https://doi.org/10.1007/s00267-016-0807-y

Niemiera, A. X., & Holle, B. V. (2009). Invasive Plant Species and the Ornamental Horticulture Industry. In Inderjit (Ed.), *Management of Invasive Weeds* (Vol. 5, pp. 167–187). Springer Netherlands. https://doi.org/10.1007/978-1-4020-9202-2_9

Nishida, T., Yamashita, N., Asai, M., Kurokawa, S., Enomoto, T., Pheloung, P. C., & Groves, R. H. (2009). Developing a pre-entry weed risk assessment system for use in Japan. *Biological Invasions*, *11*(6), 1319–1333. https://doi.org/10.1007/s10530-008-9340-0

Nishimoto, R. (2019). Global trends in the crop protection industry. *Journal of Pesticide Science*, *44*(3), 141–147. https://doi.org/10.1584/jpestics.D19-101

Njiru, M., Waithaka, E., Muchiri, M., van Knaap, M., & Cowx, I. G. (2005). Exotic introductions to the fishery of Lake Victoria: What are the management options? *Lakes & Reservoirs: Research & Management*, *10*(3), 147–155. https://doi.org/10.1111/j.1440-1770.2005.00270.x

Noar, R. D., Jahant-Miller, C. J., Emerine, S., & Hallberg, R. (2021). Early Warning Systems as a Component of Integrated Pest Management to Prevent the Introduction of Exotic Pests. *Journal of Integrated Pest Management*, *12*(1), 16. https://doi.org/10.1093/jipm/pmab011

Noble, M., Ruiz, G. M., & Murphy, K. R. (2016). Chemical Assessment of Ballast Water Exchange Compliance: Implementation in North America and New Zealand. *Frontiers in Marine Science*, *3*. https://doi.org/10.3389/fmars.2016.00066

Nogales, M., Vidal, E., Medina, F. M., Bonnaud, E., Tershy, B. R., Campbell, K. J., & Zavaleta, E. S. (2013). Feral Cats and Biodiversity Conservation: The Urgent Prioritization of Island Management. *BioScience*, *63*(10), 804–810. https://doi.org/10.1525/bio.2013.63.10.7

Nordberg, E. J., Macdonald, S., Zimny, G., Hoskins, A., Zimny, A., Somaweera, R., Ferguson, J., & Perry, J. (2019). An evaluation of nest predator impacts and the efficacy of plastic meshing on marine turtle nests on the western Cape York Peninsula, Australia. *Biological Conservation*, *238*, 108201. https://doi.org/10.1016/j.biocon.2019.108201

Nourani, S. W., Krasny, M. E., & Decker, D. J. (2018). Learning and linking for invasive species management. *Ecology and Society*, *23*(3), 29. https://doi.org/10.5751/ES-10327-230329

Novoa, A., Shackleton, R., Canavan, S., Cybèle, C., Davies, S. J., Dehnen-Schmutz, K., Fried, J., Gaertner, M., Geerts, S., Griffiths, C. L., Kaplan, H., Kumschick, S., Le Maitre, D. C., Measey, G. J., Nunes, A. L., Richardson, D. M., Robinson, T. B., Touza, J., & Wilson, J. R. U. (2018). A framework for engaging stakeholders on the management of alien species. *Journal of Environmental Management*, *205*, 286–297. https://doi.org/10.1016/j.jenvman.2017.09.059

Nsikani, M. M., van Wilgen, B. W., & Gaertner, M. (2018). Barriers to ecosystem restoration presented by soil legacy effects of invasive alien N_2-fixing woody species: Implications for ecological restoration. *Restoration Ecology*, *26*(2), 235–244. https://doi.org/10.1111/rec.12669

Nunes, A. L., Hoffman, A. C., Zengeya, T. A., Measey, G. J., & Weyl, O. L. (2017). Red swamp crayfish, *Procambarus clarkii*, found in South Africa 22 years after attempted eradication. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*(6), 1334–1340. https://doi.org/10.1002/aqc.2741

Nunes, A. L., Tricarico, E., Panov, V. E., Cardoso, A. C., & Katsanevakis, S. (2015). Pathways and gateways of freshwater invasions in Europe. *Aquatic Invasions*, *10*(4), 359–370. https://doi.org/10.3391/ai.2015.10.4.01

Nuñez, M. A., Chiuffo, M. C., Torres, A., Paul, T., Dimarco, R. D., Raal, P., Policelli, N., Moyano, J., García, R. A., van Wilgen, B. W., Pauchard, A., & Richardson, D. M. (2017). Ecology and management of invasive Pinaceae around the world: Progress and challenges. *Biological Invasions*, *19*(11), 3099–3120. https://doi.org/10.1007/s10530-017-1483-4

Nuñez, M. A., Kuebbing, S., Dimarco, R. D., & Simberloff, D. (2012). Invasive Species: To eat or not to eat, that is the question. *Conservation Letters*, *5*(5), 334–341. https://doi.org/10.1111/j.1755-263X.2012.00250.x

Nuñez, M. A., & Pauchard, A. (2010). Biological invasions in developing and developed countries: Does one model fit all? *Biological Invasions*, *12*(4), 707–714. https://doi.org/10.1007/s10530-009-9517-1

Nyangau, P., Muriithi, B., Diiro, G., Akutse, K. S., & Subramanian, S. (2020). Farmers' knowledge and management practices of cereal, legume and vegetable insect pests, and willingness to pay for biopesticides. *International Journal of Pest Management*, *68*(3), 204–216. https://doi.org/10.1080/09670874.2020.1817621

Oba, G., Byakagaba, P., & Angassa, A. (2008). Participatory monitoring of biodiversity in East African grazing lands. *Land Degradation & Development*, *19*(6), 636–648. https://doi.org/10.1002/ldr.867

O'Donnell, C. F. J., Dilks, P. J., & Elliott, G. P. (1996). Control of a stoat *(Mustela erminea*) population irruption to enhance mohua (yellowhead) (*Mohoua ochrocephala*) breeding success in New Zealand. *New Zealand Journal of Zoology*, *23*(3), 279–286. https://doi.org/10.1080/03014223.1996.9518086

OECD. (2012). *Important Issues on Risk Assessment of Manufactured Nanomaterials*. Organization for Economic Cooperation and Development, Environmental Health.

Oficialdegui, F. J., Delibes-Mateos, M., Green, A. J., Sánchez, M. I., Boyero, L., & Clavero, M. (2020). Rigid laws and invasive species management. *Conservation Biology*, *34*(4), 1047–1050. https://doi.org/10.1111/cobi.13481

Oficialdegui, F. J., Sánchez, M. I., & Clavero, M. (2020). One century away from home: How the red swamp crayfish took over the world. *Reviews in Fish Biology and Fisheries*, *30*(1), 121–135. https://doi.org/10.1007/s11160-020-09594-z

Oidtmann, B., Peeler, E., Lyngstad, T., Brun, E., Bang Jensen, B., & Stärk, K. D. C. (2013). Risk-based methods for fish and terrestrial animal disease surveillance. *Preventive Veterinary Medicine*, *112*(1–2), 13–26. https://doi.org/10.1016/j.prevetmed.2013.07.008

Olenin, S., Minchin, D., & Daunys, D. (2007). Assessment of biopollution in aquatic ecosystems. *Marine Pollution Bulletin*, *55*(7–9), 379–394. https://doi.org/10.1016/j.marpolbul.2007.01.010

Olsen, A., Konovalov, D. A., Philippa, B., Ridd, P., Wood, J. C., Johns, J., Banks, W., Girgenti, B., Kenny, O., Whinney, J., Calvert, B., Azghadi, M. R., & White, R. D. (2019). DeepWeeds: A Multiclass Weed Species Image Dataset for Deep Learning. *Scientific Reports*, *9*(1), 2058. https://doi.org/10.1038/s41598-018-38343-3

Ontario Invasive Plant Council. (2021). Best Management Practices. *Ontario Invasive Plant Council*. https://www.ontarioinvasiveplants.ca/resources/best-management-practices/

Oppel, S., Beaven, B. M., Bolton, M., Vickery, J., & Bodey, T. W. (2011). Eradication of Invasive Mammals on Islands Inhabited by Humans and Domestic Animals. *Conservation Biology*, *25*(2), 232–240. https://doi.org/10.1111/j.1523-1739.2010.01601.x

Orapa, W. (2017). Impact and management of invasive alien plants in Pacific Island communities. In C. A. Ellison, K. V. Sankaran, & S. T. Murphy (Eds.), *Invasive alien plants: Impacts on development and options for management* (pp. 73–108). CABI. https://doi.org/10.1079/9781780646275.0073

Orr, A., Mwale, B., & Saiti, D. (2002). Modelling agricultural "performance": Smallholder weed management in Southern Malawi. *International Journal of Pest Management*, *48*(4), 265–278. https://doi.org/10.1080/09670870210149808

Ortiz, M., Rodriguez-Zaragoza, F., Hermosillo-Nuñez, B., & Jordán, F. (2015). Control Strategy Scenarios for the Alien Lionfish *Pterois volitans* in Chinchorro Bank (Mexican Caribbean): Based on Semi-Quantitative Loop Analysis. *PLoS ONE*, *10*(6), e0130261. https://doi.org/10.1371/journal.pone.0130261

Osunkoya, O. O., Froese, J. G., & Nicol, S. (2019). Management feasibility of established invasive plant species in Queensland, Australia: A stakeholders' perspective. *Journal of Environmental Management*, *246*, 484–495. https://doi.org/10.1016/j.jenvman.2019.05.052

Oswald, A. (2005). *Striga* control—Technologies and their dissemination. *Crop Protection*, *24*(4), 333–342. https://doi.org/10.1016/j.cropro.2004.09.003

Otim, M. H., Tay, W. T., Walsh, T. K., Kanyesigye, D., Adumo, S., Abongosi, J., Ochen, S., Sserumaga, J., Alibu, S., Abalo, G., Asea, G., & Agona, A. (2018). Detection of sister-species in invasive populations of the fall armyworm *Spodoptera frugiperda* (Lepidoptera: Noctuidae) from Uganda. *PLoS ONE*, *13*(4), e0194571. https://doi.org/10.1371/journal.pone.0194571

Owen, S. J., & Sheldon, J. K. (1996). Strategies for ecological weed control on conservation lands in New Zealand. *Proceedings of the Eleventh Australian Weeds Conference*, *2*(5). https://caws.org.nz/old-site/awc/1996/awc199615161.pdf

Paap, T., Burgess, T. I., & Wingfield, M. J. (2017). Urban trees: Bridge-heads for forest pest invasions and sentinels for early detection. *Biological Invasions*, *19*(12), 3515–3526. https://doi.org/10.1007/s10530-017-1595-x

Paap, T., Wingfield, M. J., de Beer, Z. W., & Roets, F. (2020). Lessons from a major pest invasion: The polyphagous shot hole borer in South Africa. *South African Journal of Science*, *116*(11/12). https://doi.org/10.17159/sajs.2020/8757

Padayachee, A. L., Irlich, U. M., Faulkner, K. T., Gaertner, M., Procheş, Ş., Wilson, J. R. U., & Rouget, M. (2017). How do invasive species travel to and through urban environments? *Biological Invasions*, *19*(12), 3557–3570. https://doi.org/10.1007/s10530-017-1596-9

Page, A. R., & Lacey, K. L. (2006). *Economic Impact Assessment of Australian Weed Biological Control: Report to the CRC for Australian Weed Management*. CRC for Australian Weed Management. https://books.google.com.au/books?id=LrHkAAAACAAJ

Paini, D. R., Sheppard, A. W., Cook, D. C., De Barro, P. J., Worner, S. P., & Thomas, M. B. (2016). Global threat to agriculture from invasive species. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(27), 7575–7579. https://doi.org/10.1073/pnas.1602205113

Pan, H., Xu, L., Noland, J. E., Li, H., Siegfried, B. D., & Zhou, X. (2016). Assessment of Potential Risks of Dietary RNAi to a Soil Micro-arthropod, *Sinella curviseta* Brook (Collembola: Entomobryidae). *Frontiers in Plant Science*, *7*, 1028. https://doi.org/10.3389/fpls.2016.01028

Panetta, F. D. (2012). Evaluating the performance of weed containment programmes. *Diversity and Distributions*, *18*(10), 1024–1032. https://doi.org/10.1111/j.1472-4642.2012.00932.x

Panetta, F. D., Cacho, O., Hester, S., Sims-Chilton, N., & Brooks, S. (2011). Estimating and influencing the duration of weed eradication programmes. *Journal of Applied Ecology*, *48*(4), 980–988. https://doi.org/10.1111/j.1365-2664.2011.02000.x

Panetta, F. D., & Cacho, O. J. (2014). Designing weed containment strategies: An approach based on feasibilities of eradication and containment. *Diversity and Distributions*, *20*(5), 555–566. https://doi.org/10.1111/ddi.12170

Panzacchi, M., Cocchi, R., Genovesi, P., & Bertolino, S. (2007). Population control of coypu *Myocastor coypus* in Italy compared to eradication in UK: A cost-benefit analysis. *Wildlife Biology*, *13*(2), 159–171. https://doi.org/10.2981/0909-6396(2007)13[159:PCOCMC]2.0.CO;2

Papadimitriou, V. C., Portmann, R. W., Fahey, D. W., Mühle, J., Weiss, R. F., & Burkholder, J. B. (2008). Experimental and Theoretical Study of the Atmospheric Chemistry and Global Warming Potential of SO_2F_2. *The Journal of Physical Chemistry A*, *112*(49), 12657–12666. https://doi.org/10.1021/jp806368u

Park, E., Loc, H. H., Van Binh, D., & Kantoush, S. (2022). The worst 2020 saline water intrusion disaster of the past century in the Mekong Delta: Impacts, causes, and management implications. *Ambio*, *51*(3), 691–699. https://doi.org/10.1007/s13280-021-01577-z

Park, K. (2004). Assessment and Management of Invasive Alien Predators. *Ecology and Society*, *9*(2), 12. https://doi.org/10.5751/ES-01208-090212

Parkes, J., Fisher, P., Robinson, S., & Aguirre-Muñoz, A. (2014). Eradication of feral cats from large islands: An assessment of the effort required for success. *New Zealand Journal of Ecology*, *38*(2), 307–314. https://www.jstor.org/stable/24060808

Parrot, D., Roy, S., Baker, R., Cannon, R., Eyre, D., Hill, M., Wagner, M., Preston, C., Roy, H., Beckmann, B., Copp, G. H., Edmonds, N., Ellis, J., Laing, I., Britton, J. R., Gozlan, R. E., & Mumford, J. (2009). *Horizon scanning for new invasive non-native animal species in England* (Natural England Commissioned Report NECR009; p. 121). Natural England. http://publications.naturalengland.org.uk/publication/43005

Paschoal, A. M. O., Massara, R. L., Bailey, L. L., Kendall, W. L., Doherty, P. F., Hirsch, A., Chiarello, A. G., & Paglia, A. P. (2016). Use of Atlantic Forest protected areas by free-ranging dogs: Estimating abundance and persistence of use. *Ecosphere*, *7*(10), e01480. https://doi.org/10.1002/ecs2.1480

Pauchard, A., & Alaback, P. B. (2004). Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile. *Conservation Biology*, *18*(1), 238–248. https://doi.org/10.1111/j.1523-1739.2004.00300.x

Paudel, S., Mansfield, S., Villamizar, L. F., Jackson, T. A., & Marshall, S. D. G. (2021). Can Biological Control Overcome the Threat From Newly Invasive Coconut Rhinoceros Beetle Populations (Coleoptera: Scarabaeidae)? A Review. *Annals of the Entomological Society of America*, *114*(2), 247–256. https://doi.org/10.1093/aesa/saaa057

Paul, A., Radhakrishnan, M., Anandakumar, S., Shanmugasundaram, S., & Anandharamakrishnan, C. (2020). Disinfestation techniques for major cereals: A status report. *Comprehensive Reviews in Food Science and Food Safety*, *19*(3), 1125–1155. https://doi.org/10.1111/1541-4337.12555

Paula, A. F., & Creed, J. C. (2005). Spatial distribution and abundance of nonindigenous coral genus *Tubastraea* (*Cnidaria, Scleractinia*) around Ilha Grande, Brazil. *Brazilian Journal of Biology*, *65*(4), 661–673. https://doi.org/10.1590/S1519-69842005000400014

Pawson, S. M., Sullivan, J. J., & Grant, A. (2020). Expanding general surveillance of invasive species by integrating citizens as both observers and identifiers. *Journal of Pest Science*, *93*, 1155–1166. https://doi.org/10.1007/s10340-020-01259-x

Payne, P. R., Finlay-Smits, S., Small, B., Cave, V., & Kean, J. (2021). *What's That Bug? Citizen Science For Biosecurity In Mount Maunganui, New Zealand* [Preprint]. In Review. https://doi.org/10.21203/rs.3.rs-746797/v1

Paynter, Q., Overton, J. M., Hill, R. L., Bellgard, S. E., & Dawson, M. I. (2012). Plant traits predict the success of weed biocontrol. *Journal of Applied Ecology*, *49*(5), 1140–1148. https://doi.org/10.1111/j.1365-2664.2012.02178.x

Peacock, L., Twaddle, M., Craddock, P., & Nagle, D. (2019). National Invasive Ant Surveillance Programme. *Surveillance*, *46*(3 Annual Report), 81–82. http://www.sciquest.org.nz/node/157807

Pearson, D. E., & Callaway, R. M. (2005). Indirect nontarget effects of host-specific biological control agents: Implications for biological control. *Biological Control*, *35*(3), 288–298. https://doi.org/10.1016/j.biocontrol.2005.05.011

Pearson, D. E., Ortega, Y. K., Runyon, J. B., & Butler, J. L. (2016). Secondary invasion: The bane of weed management. *Biological Conservation*, *197*, 8–17. https://doi.org/10.1016/j.biocon.2016.02.029

Pejchar, L., & Mooney, H. A. (2009). Invasive species, ecosystem services and human well-being. *Trends in Ecology & Evolution*, *24*(9), 497–504. https://doi.org/10.1016/j.tree.2009.03.016

Peltzer, D. A., Bellingham, P. J., Dickie, I. A., Houliston, G., Hulme, P. E., Lyver, P. O., McGlone, M., Richardson, S. J., & Wood, J. (2019). Scale and complexity implications of making New Zealand predator-free by 2050. *Journal of the Royal Society of New Zealand*, *49*(3), 412–439. https://doi.org/10.1080/03036758.2019.1653940

Pergl, J., Härtel, H., Pyšek, P., & Stejskal, R. (2020). Don't throw the baby out with the bathwater–ban of glyphosate use depends on context. *NeoBiota*, *56*, 27–29. https://doi.org/10.3897/neobiota.56.51823

Pergl, J., Pyšek, P., Bacher, S., Essl, F., Genovesi, P., Harrower, C. A., Hulme, P. E., Jeschke, J. E., Kenis, M., Kühn, I., Perglová, I., Rabitsch, W., Roques, A., Roy, D. B., Roy, H. E., Vilà, M., Winter, M., & Nentwig, W. (2017). Troubling travellers: Are ecologically harmful alien species associated with particular introduction pathways? *NeoBiota*, *32*(October 2016), 1–20. https://doi.org/10.3897/neobiota.32.10199

Perrings, C. (2016). Options for managing the infectious animal and plant disease risks of international trade. *Food Security*, *8*(1), 27–35. https://doi.org/10.1007/s12571-015-0523-0

Perroy, R. L., Sullivan, T., & Stephenson, N. (2017). Assessing the impacts of canopy openness and flight parameters on detecting a sub-canopy tropical invasive plant using a small unmanned aerial system. *ISPRS Journal of Photogrammetry and Remote Sensing*, *125*, 174–183. https://doi.org/10.1016/j.isprsjprs.2017.01.018

Pettorelli, N., Laurance, W. F., O'Brien, T. G., Wegmann, M., Nagendra, H., & Turner, W. (2014). Satellite remote sensing for applied ecologists: Opportunities and challenges.

Journal of Applied Ecology, *51*(4), 839–848. https://doi.org/10.1111/1365-2664.12261

Petucco, C., Lobianco, A., & Caurla, S. (2020). Economic Evaluation of an Invasive Forest Pathogen at a Large Scale: The Case of Ash Dieback in France. *Environmental Modeling & Assessment*, *25*(1), 1–21. https://doi.org/10.1007/s10666-019-09661-1

Peyre, M., Haesler, B., Hoinville, L., Goutard, F., Cameron, A., Dorea, F., Traon, D., Grosbois, V., & Schauer, B. (2019). *The EVA tool: A decision support tool for the evaluation of surveillance systems*. RiskSur. https://www.fp7-risksur.eu/

Peyton, J., Martinou, A. F., Adriaens, T., Chartosia, N., Karachle, P. K., Rabitsch, W., Tricarico, E., Arianoutsou, M., Bacher, S., Bazos, I., Brundu, G., Bruno-McClung, E., Charalambidou, I., Demetriou, M., Galanidi, M., Galil, B., Guillem, R., Hadjiafxentis, K., Hadjioannou, L., … Roy, H. E. (2020). Horizon Scanning to Predict and Prioritize Invasive Alien Species With the Potential to Threaten Human Health and Economies on Cyprus. *Frontiers in Ecology and Evolution*, *8*. https://doi.org/10.3389/fevo.2020.566281

Peyton, J., Martinou, A. F., Pescott, O. L., Demetriou, M., Adriaens, T., Arianoutsou, M., Bazos, I., Bean, C. W., Booy, O., Botham, M., Britton, J. R., Cervia, J. L., Charilaou, P., Chartosia, N., Dean, H. J., Delipetrou, P., Dimitriou, A. C., Dörflinger, G., Fawcett, J., … Roy, H. E. (2019). Horizon scanning for invasive alien species with the potential to threaten biodiversity and human health on a Mediterranean island. *Biological Invasions*, *21*(6), 2107–2125. https://doi.org/10.1007/s10530-019-01961-7

Pfeiffer, J. M., & Voeks, R. A. (2008). Biological invasions and biocultural diversity: Linking ecological and cultural systems. *Environmental Conservation*, *35*(4), 281–293. https://doi.org/10.1017/S0376892908005146

Pheloung, P. C., Williams, P. A., & Halloy, S. R. (1999). A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. *Journal of Environmental Management*, *57*(4), 239–251. https://doi.org/10.1006/jema.1999.0297

Phillips McDougall. (2016). *The Cost of New Agrochemical Product Discovery, Development and Registration in 1995, 2000, 2005-8 and 2010-2014. R&D expenditure in 2014 and expectations for 2019*. https://croplife.org/wp-content/uploads/2016/04/Cost-of-CP-report-FINAL.pdf

Phillips McDougall. (2018). *Evolution of the Crop Protection Industry since 1960*. https://croplife.org/wp-content/uploads/2018/11/Phillips-McDougall-Evolution-of-the-Crop-Protection-Industry-since-1960-FINAL.pdf

Phillips, R. B., Wiedenfeld, D. A., & Snell, H. L. (2012). Current status of alien vertebrates in the Galápagos Islands: Invasion history, distribution, and potential impacts. *Biological Invasions*, *14*(2), 461–480. https://doi.org/10.1007/s10530-011-0090-z

Pickart, A. (2012). *Spartina densiflora* invasion ecology and the restoration of native salt marshes, Huumboldt Bay, California. *Unpublished Report, US Fish & Wildlife Service, Arcata, California*. https://www.nfwf.org/sites/default/files/finalreports1/25318_FinalReport.pdf

Pickering, C., & Mount, A. (2010). Do tourists disperse weed seed? A global review of unintentional human-mediated terrestrial seed dispersal on clothing, vehicles and horses. *Journal of Sustainable Tourism*, *18*(2), 239–256. https://doi.org/10.1080/09669580903406613

Piel, F., Gilbert, M., De Cannière, C., & Grégoire, J.-C. (2007). Coniferous round wood imports from Russia and Baltic countries to Belgium. A pathway analysis for assessing risks of exotic pest insect introductions. *Diversity and Distributions*, *14*(2), 318–328. https://doi.org/10.1111/j.1472-4642.2007.00390.x

Piertney, S. B., Black, A., Watt, L., Christie, D., Poncet, S., & Collins, M. A. (2016). Resolving patterns of population genetic and phylogeographic structure to inform control and eradication initiatives for brown rats *Rattus norvegicus* on South Georgia. *Journal of Applied Ecology*, *53*(2), 332–339. https://doi.org/10.1111/1365-2664.12589

Pimentel, D., Acquay, H., Biltonen, M., Rice, P., Silva, M., Nelson, J., Lipner, V., Giordano, S., Horowitz, A., & D'Amore, M. (1992). Environmental and Economic Costs of Pesticide Use. *BioScience*, *42*(10), 750–760. https://doi.org/10.2307/1311994

Pimentel, D., & Andow, D. A. (1984). Pest management and pesticide impacts. *International Journal of Tropical Insect Science*, *5*(3), 141–149. https://doi.org/10.1017/S1742758400008201

Pimentel, M. A. G., Faroni, L. R. D., Guedes, R. N. C., Sousa, A. H., & Tótola, M. R. (2009). Phosphine resistance in Brazilian populations of *Sitophilus zeamais* Motschulsky (Coleoptera: Curculionidae). *Journal of Stored Products Research*, *45*(1), 71–74. https://doi.org/10.1016/j.jspr.2008.09.001

Piria, M., Copp, G. H., Dick, J. T. A., Duplić, A., Groom, Q., Jelić, D., Lucy, F. E., Roy, H. E., Sarat, E., Simonović, P., Tomljanović, T., Tricarico, E., Weinlander, M., Adámek, Z., Sarah Bedolfe, Coughlan, N. E., Davis, E., Dobrzycka-Krahel, A., Grgić, Z., … Caffrey, J. M. (2017). Tackling invasive alien species in Europe II: Threats and opportunities until 2020. *Management of Biological Invasions*, *8*(3), 273–286. https://doi.org/10.3391/mbi.2017.8.3.02

Pluess, T., Cannon, R., Jarošík, V., Pergl, J., Pyšek, P., & Bacher, S. (2012). When are eradication campaigns successful? A test of common assumptions. *Biological Invasions*, *14*(7), 1365–1378. https://doi.org/10.1007/s10530-011-0160-2

Pluess, T., Jarošík, V., Pyšek, P., Cannon, R., Pergl, J., Breukers, A., & Bacher, S. (2012). Which Factors Affect the Success or Failure of Eradication Campaigns against Alien Species? *PLoS ONE*, *7*(10), e48157. https://doi.org/10.1371/journal.pone.0048157

Poland, T. M., & McCullough, D. G. (2006). Emerald Ash Borer: Invasion of the Urban Forest and the Threat to North America's Ash Resource. *Journal of Forestry*, *104*(3), 118–124. https://doi.org/10.1093/jof/104.3.118

Poncio, L. C., Anjos, F. A. dos, Oliveira, D. A. de, Rebechi, D., Oliveira, R. N. de, Chitolina, R. F., Fermino, M. L., Bernardes, L. G., Guimarães, D., Lemos, P. A., Silva, M. N. E., Silvestre, R. G. M., Bernardes, E. S., & Paldi, N. (2019). *Successful suppression of a field population of* Ae. Aegypti *mosquitoes using a novel biological vector control strategy is associated with significantly lower incidence of dengue* (p. 19010678). medRxiv. https://doi.org/10.1101/19010678

Popay, I., & Field, R. (1996). Grazing Animals as Weed Control Agents. *Weed Technology*, *10*(1), 217–231. https://doi.org/10.1017/S0890037X00045942

Potamitis, I., & Rigakis, I. (2015). *Smart traps for automatic remote monitoring of* Rhynchophorus ferrugineus *(Coleoptera: Curculionidae)* (e1651). PeerJ Inc. https://doi.org/10.7287/peerj.preprints.1337v1

PREDATOR Free NZ. (2021). *Home*. Predator Free NZ Trust. https://predatorfreenz.org/

Preti, M., Verheggen, F., & Angeli, S. (2021). Insect pest monitoring with camera-equipped traps: Strengths and limitations. *Journal of Pest Science*, *94*(2), 203–217. https://doi.org/10.1007/s10340-020-01309-4

Pretty Paint-Small, V. (2013). *Linking culture, ecology and policy: The invasion of Russian-olive (*Elaeagnus angustifolia *l.) on the Crow Indian Reservation, south-central Montana, USA* [Thesis, Colorado State University. Libraries]. https://mountainscholar.org/handle/10217/78865

Priono, B., & Satyani, D. (2010). Sekilas Tentang Beberapa Jenis Ikan Hias Air Tawar Yang Dilarang Masuk Ke Indonesia. *Media Akuakultur*, *5*(2), 102–108. https://doi.org/10.15578/ma.5.2.2010.102-108

Prior, K. M., Adams, D. C., Klepzig, K. D., & Hulcr, J. (2018). When does invasive species removal lead to ecological recovery? Implications for management success. *Biological Invasions*, *20*(2), 267–283.

Probert, A. F., Volery, L., Kumschick, S., Vimercati, G., & Bacher, S. (2020). Understanding uncertainty in the Impact Classification for Alien Taxa (ICAT) assessments. *NeoBiota*, *62*, 387–405. https://doi.org/10.3897/neobiota.62.52010

Probert, A. F., Wegmann, D., Volery, L., Adriaens, T., Bakiu, R., Bertolino, S., Essl, F., Gervasini, E., Groom, Q., Latombe, G., Marisavljevic, D., Mumford, J., Pergl, J., Preda, C., Roy, H. E., Scalera, R., Teixeira, H., Tricarico, E., Vanderhoeven, S., & Bacher, S. (2022). Identifying, reducing, and communicating uncertainty in community science: A focus on alien species. *Biological Invasions*, *24*(11), 3395–3421. https://doi.org/10.1007/s10530-022-02858-8

Proctor, W., & Drechsler, M. (2006). Deliberative Multicriteria Evaluation. *Environment and Planning C: Government and Policy*, *24*(2), 169–190. https://doi.org/10.1068/c22s

Provencher, B., Lewis, D. J., & Anderson, K. (2012). Disentangling preferences and expectations in stated preference analysis with respondent uncertainty: The case of invasive species prevention. *Journal of Environmental Economics and Management*, *64*(2), 169–182. https://doi.org/10.1016/j.jeem.2012.04.002

Provencher, L., Forbis, T. A., Frid, L., & Medlyn, G. (2007). Comparing alternative management strategies of fire, grazing, and weed control using spatial modeling. *Ecological Modelling*, *209*(2), 249–263. https://doi.org/10.1016/j.ecolmodel.2007.06.030

Pyšek, P., Hulme, P. E., Meyerson, L. A., Smith, G. F., Boatwright, J. S., Crouch, N. R., Figueiredo, E., Foxcroft, L. C., Jarošík, V., Richardson, D. M., Suda, J., & Wilson, J. R. U. (2013). Hitting the right target: Taxonomic challenges for, and of, plant invasions. *AoB PLANTS*, *5*, plt042. https://doi.org/10.1093/aobpla/plt042

Pyšek, P., Hulme, P. E., Simberloff, D., Bacher, S., Blackburn, T. M., Carlton, J. T., Dawson, W., Essl, F., Foxcroft, L. C., Genovesi, P., Jeschke, J. M., Kühn, I., Liebhold, A. M., Mandrak, N. E., Meyerson, L. A., Pauchard, A., Pergl, J., Roy, H. E., Seebens, H., … Richardson, D. M. (2020). Scientists' warning on invasive alien species. *Biological Reviews*, *95*(6), 1511–1534. https://doi.org/10.1111/brv.12627

Rai, R. K., & Scarborough, H. (2013). Economic value of mitigation of plant invaders in a subsistence economy: Incorporating labour as a mode of payment. *Environment and Development Economics*, *18*(2), 225–244. https://doi.org/10.1017/S1355770X1200037X

Raine, A. F., Vynne, M., & Driskill, S. (2019). The impact of an introduced avian predator, the Barn Owl *Tyto alba*, on Hawaiian seabirds. *Marine Ornithology*, *47*, 33–38.

Rainwater-Lovett, K., Pacheco, J. M., Packer, C., & Rodriguez, L. L. (2009). Detection of foot-and-mouth disease virus infected cattle using infrared thermography. *The Veterinary Journal*, *180*(3), 317–324. https://doi.org/10.1016/j.tvjl.2008.01.003

Ramsar Convention on Wetlands. (2018). *Global Wetland Outlook: State of the World's Wetlands and their Services to People.* Ramsar Convention Secretariat.

Ramsar Convention Secretariat. (2010). *Managing wetlands: Frameworks for managing Wetlands of International Importance and other wetland sites* (4th ed., Vol. 18). Ramsar Convention Secretariat. https://www.ramsar.org/sites/default/files/documents/pdf/lib/hbk4-18.pdf

Ramsey, D. S. L., Parkes, J., & Morrison, S. A. (2009). Quantifying Eradication Success: The Removal of Feral Pigs from Santa Cruz Island, California. *Conservation Biology*, *23*(2), 449–459. https://doi.org/10.1111/j.1523-1739.2008.01119.x

Ramsey, D. S. L., Parkes, J. P., Will, D., Hanson, C. C., & Campbell, K. J. (2011). Quantifying the success of feral cat eradication, San Nicolas Island, California. *New Zealand Journal of Ecology*, *35*(2), 163–173. https://www.jstor.org/stable/24060664

Randall, J. M. (1996). Weed Control for the Preservation of Biological Diversity. *Weed Technology*, *10*(2), 370–383. https://doi.org/10.1017/S0890037X00040124

Ranjan, R. (2019). Deriving double dividends through linking payments for ecosystem services to environmental entrepreneurship: The case of the invasive weed *Lantana camara*. *Ecological Economics*, *164*, 106380. https://doi.org/10.1016/j.ecolecon.2019.106380

Reaser, J. K. (2003). Invasive alien species: Definitions, causes, consequences, and the way forward. In N. Pallewatta, J. K. Reaser, & A. Gutierrez (Eds.), *Prevention and Management of Invasive Alien Species: Proceedings of a Workshop on Forging Cooperation throughout South and Southeast Asia* (pp. 11–18). Global Invasive Species Programme. https://giasipartnership.myspecies.info/en/content/pallewatta-n-jk-reaser-gutierrez-eds-2003-prevention-and-management-invasive-alien-species

Reaser, J. K., Burgiel, S. W., Kirkey, J., Brantley, K. A., Veatch, S. D., & Burgos-Rodríguez, J. (2020). The early detection of and rapid response (EDRR) to invasive species: A conceptual framework and federal capacities assessment. *Biological Invasions*, *22*(1), 1–19. https://doi.org/10.1007/s10530-019-02156-w

Reaser, J. K., Simpson, A., Guala, G. F., Morisette, J. T., & Fuller, P. (2020). Envisioning a national invasive species information framework. *Biological Invasions*, *22*(1), 21–36. https://doi.org/10.1007/s10530-019-02141-3

Rebelo, M. F., Afonso, L. F., Americo, J. A., da Silva, L., Neto, J. L. B., Dondero, F., & Zhang, Q. (2018). A sustainable synthetic biology approach for the control of the invasive golden mussel (*Limnoperna fortunei*). *PeerJ Preprints*, *6*, e27164v3. https://doi.org/10.7287/peerj.preprints.27164v3

Redford, K. H., Brooks, T. M., Macfarlane, N. B. W., & Adams, J. S. (Eds.). (2019). *Genetic frontiers for conservation: An assessment of synthetic biology and biodiversity conservation: technical assessment*. IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2019.05.en

Rees, H. C., Maddison, B. C., Middleditch, D. J., Patmore, J. R. M., & Gough, K. C. (2014). REVIEW: The detection of aquatic animal species using environmental DNA – a review of eDNA as a survey tool in ecology. *Journal of Applied Ecology*, *51*(5), 1450–1459. https://doi.org/10.1111/1365-2664.12306

Regan, H. M., Colyvan, M., & Burgman, M. A. (2002). A Taxonomy and Treatment of Uncertainty for Ecology and Conservation Biology. *Ecological Applications*, *12*(2), 618–628. https://doi.org/10.1890/1051-0761(2002)012[0618:ATATOU]2.0.CO;2

Regan, T. J., McCarthy, M. A., Baxter, P. W. J., Panetta, F. D., & Possingham, H. P. (2006). Optimal eradication: When to stop looking for an invasive plant. *Ecology Letters*, *9*(7), 759–766. https://doi.org/10.1111/j.1461-0248.2006.00920.x

Reichard, S. H., & Hamilton, C. W. (1997). Predicting Invasions of Woody Plants Introduced into North America. *Conservation Biology*, *11*(1), 193–203. https://doi.org/10.1046/j.1523-1739.1997.95473.x

Reisen, W., Lothrop, H., Chiles, R., Madon, M., Cossen, C., Woods, L., Husted, S., Kramer, V., & Edman, J. (2004). West Nile Virus in California. *Emerging Infectious Diseases*, *10*(8), 1369–1378. https://doi.org/10.3201/eid1008.040077

Rejmanek, M., & Pitcairn, M. J. (2002). When is eradication of exotic pest plants a realistic goal? In C. R. Veitch & M. N. Clout (Eds.), *Turning the tide: The eradication of invasive species: Proceedings of the International Conference on eradication of Island Invasives* (pp. 249–253). IUCN. https://portals.iucn.org/library/efiles/documents/ssc-op-028.pdf

Rettberg, S. (2010). Contested narratives of pastoral vulnerability and risk in Ethiopia's Afar region. *Pastoralism – Research, Policy and Practice*, *1*(2), 248–273. https://doi.org/10.3362/2041-7136.2010.014

Rettberg, S., & Müller-Mahn, D. (2012). Human Environment Interactions: The invasion of *Prosopis juliflora* in the Drylands of Northeast Ethiopia. In L. Mol & T. Sternberg (Eds.), *Changing Deserts – Integrating People and their Environments.* (pp. 297-316.). Whitehorse Press. http://dx.doi.org/10.3197/9781912186310.ch15

Reyes-García, V., Fernández-Llamazares, Á., McElwee, P., Molnár, Z., Öllerer, K., Wilson, S. J., & Brondizio, E. S. (2019). The contributions of Indigenous Peoples and local communities to ecological restoration. *Restoration Ecology*, *27*(1), 3–8. https://doi.org/10.1111/rec.12894

Ricciardi, A., Blackburn, T. M., Carlton, J. T., Dick, J. T. A., Hulme, P. E., Iacarella, J. C., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Sutherland, W. J., Wardle, D. A., & Aldridge, D. C. (2017). Invasion Science: A Horizon Scan of Emerging Challenges and Opportunities. *Trends in Ecology & Evolution*, *32*(6), 464–474. https://doi.org/10.1016/j.tree.2017.03.007

Ricciardi, A., Iacarella, J. C., Aldridge, D. C., Blackburn, T. M., Carlton, J. T., Catford, J. A., Dick, J. T. A., Hulme, P. E., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Meyerson, L. A., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Vilà, M., & Wardle, D. A. (2020). Four priority areas to advance invasion science in the face of rapid environmental change. *Environmental Reviews*, *29*(2), 119–141. https://doi.org/10.1139/er-2020-0088

Riccò, M., Ferraro, P., Gualerzi, G., Ranzieri, S., Henry, B. M., Said, Y. B., Pyatigorskaya, N. V., Nevolina, E., Wu, J., Bragazzi, N. L., & Signorelli, C. (2020). Point-of-Care Diagnostic Tests for Detecting SARS-CoV-2 Antibodies: A Systematic Review and Meta-Analysis of Real-World Data. *Journal of Clinical Medicine*, *9*(5), 1515. https://doi.org/10.3390/jcm9051515

Richardson, D. M., & van Wilgen, B. W. (2004). Invasive alien plants in South Africa: How well do we understand the ecological impacts?: working for water. *South African Journal of Science*, *100*(1), 45–52. https://doi.org/10.10520/EJC96214

Ridpath, M. G., & Waithman, J. (1988). Controlling Feral Asian Water Buffalo in Australia. *Wildlife Society Bulletin (1973-2006)*, *16*(4), 385–390. https://www.jstor.org/stable/3782438

Ringma, J., Legge, S., Woinarski, J., Radford, J., Wintle, B., & Bode, M. (2018). Australia's mammal fauna requires a strategic and enhanced network of predator-free havens. *Nature Ecology & Evolution*, *2*(3), 410–411. https://doi.org/10.1038/s41559-017-0456-4

Robertson, P. A., Adriaens, T., Caizergues, A., Cranswick, P. A., Devos, K., Gutiérrez-Expósito, C., Henderson, I., Hughes, B., Mill, A. C., & Smith, G. C. (2015). Towards the European eradication of the North American ruddy duck. *Biological Invasions*, *17*(1), 9–12. https://doi.org/10.1007/s10530-014-0704-3

Robertson, P. A., Adriaens, T., Lambin, X., Mill, A., Roy, S., Shuttleworth, C. M., & Sutton-Croft, M. (2017). The large-scale removal of mammalian invasive alien species in Northern Europe. *Pest Management Science*, *73*(2), 273–279. https://doi.org/10.1002/ps.4224

Robertson, P. A., Mill, A., Novoa, A., Jeschke, J. M., Essl, F., Gallardo, B., Geist, J., Jarić, I., Lambin, X., Musseau, C., Pergl, J., Pyšek, P., Rabitsch, W., von Schmalensee, M., Shirley, M., Strayer, D. L., Stefansson, R. A., Smith, K., & Booy, O. (2020). A proposed unified framework to describe the management of biological invasions. *Biological Invasions*, *22*(9), 2633–2645. https://doi.org/10.1007/s10530-020-02298-2

Robertson, P. A., Roy, S., Mill, A. C., Shirley, M., Adriaens, T., Ward, A. I., Tatayah, V., & Booy, O. (2019). Invasive species removals and scale – contrasting island and mainland experience. *Island Invasives: Scaling up to Meet the Challenge.: Proceedings of the International Conference on Island Invasives.*, *62*, 687–691. https://doi.org/10.2305/IUCN.CH.2019.SSC-OP.62.en

Robinson, A. P., Walshe, T., Burgman, M. A., & Nunn, M. (Eds.). (2017). *Invasive Species: Risk Assessment and Management*. Cambridge University Press. https://doi.org/10.1017/9781139019606

Robinson, A. S. (2002). Genetic Sexing Strains in Medfly, *Ceratitis capitata*, Sterile Insect Technique Programmes. *Genetica*, *116*(1), 5–13. https://doi.org/10.1023/A:1020951407069

Robinson, T. P., van Klinken, R. D., & Metternicht, G. (2010). Comparison of alternative strategies for invasive species distribution modeling. *Ecological Modelling*, *221*(19), 2261–2269. https://doi.org/10.1016/j.ecolmodel.2010.04.018

Rocamora, G. (2015). *Biosecurity protocols for protected areas and islands of high biodiversity value in Seychelles*.

Roche, R. C., Monnington, J. M., Newstead, R. G., Sambrook, K., Griffith, K., Holt, R. H. F., & Jenkins, S. R. (2015). Recreational vessels as a vector for marine non-natives: Developing biosecurity measures and managing risk through an in-water encapsulation system. *Hydrobiologia*, *750*(1), 187–199. https://doi.org/10.1007/s10750-014-2131-y

Rolheiser, K. C., Dunham, A., Switzer, S. E., Pearce, C. M., & Therriault, T. W. (2012). Assessment of chemical treatments for controlling *Didemnum vexillum*, other biofouling, and predatory sea stars in Pacific oyster aquaculture. *Aquaculture*, *364–365*, 53–60. https://doi.org/10.1016/j.aquaculture.2012.07.038

Rondeau, D. (2001). Along the Way Back from the Brink. *Journal of Environmental Economics and Management*, *42*(2), 156–182. https://doi.org/10.1006/jeem.2000.1157

Rout, T. M., Kirkwood, R., Sutherland, D. R., Murphy, S., & McCarthy, M. A. (2014). When to declare successful eradication of an invasive predator? *Animal Conservation*, *17*(2), 125–132. https://doi.org/10.1111/acv.12065

Rout, T. M., Salomon, Y., & McCarthy, M. A. (2009). Using sighting records to declare eradication of an invasive species. *Journal of Applied Ecology*, *46*(1), 110–117. https://doi.org/10.1111/j.1365-2664.2008.01586.x

Roux, D. J., & Foxcroft, L. C. (2011). The development and application of strategic adaptive management within South African National Parks. *Koedoe*, *53*(2), #1049. https://doi.org/10.4102/koedoe. v53i2.1049

Roy, H. E., Bacher, S., Essl, F., Adriaens, T., Aldridge, D. C., Bishop, J. D. D., Blackburn, T. M., Branquart, E., Brodie, J., Carboneras, C., Cottier-Cook, E. J., Copp, G. H., Dean, H. J., Eilenberg, J., Gallardo, B., Garcia, M., García-Berthou, E., Genovesi, P., Hulme, P. E., … Rabitsch, W. (2018). Developing a list of invasive alien species likely to threaten biodiversity and ecosystems in the European Union. *Global Change Biology*, *25*(3), 1032–1048. https://doi.org/10.1111/gcb.14527

Roy, H. E., Brown, P. M. J., Adriaens, T., Berkvens, N., Borges, I., Clusella-Trullas, S., Comont, R. F., De Clercq, P., Eschen, R., Estoup, A., Evans, E. W., Facon, B., Gardiner, M. M., Gil, A., Grez, A. A., Guillemaud, T., Haelewaters, D., Herz, A., Honek, A., … Zhao, Z. (2016). The harlequin ladybird, *Harmonia axyridis*: Global perspectives on invasion history and ecology. *Biological Invasions*, *18*(4), 997–1044. https://doi.org/10.1007/s10530-016-1077-6

Roy, H. E., Peyton, J. M., Aldridge, D. C., Bantock, T., Blackburn, T. M., Britton, R., Clark, P., Cook, E., Dehnen-Schmutz, K., Dines, T., Dobson, M., Edwards, F., Harrower, C., Harvey, M. C., Minchin, D., Noble, D. G., Parrott, D., Pocock, M. J. O., Preston, C. D., … Walker, K. J. (2014). Horizon scanning for invasive alien species with the potential to threaten biodiversity in Great Britain. *Global Change Biology*, *20*(12), 3859–3871. https://doi.org/10.1111/gcb.12603

Roy, H. E., Peyton, J. M., & Booy, O. (2020). Guiding principles for utilizing social influence within expert-elicitation to inform conservation decision-making. *Global Change Biology*, *26*(6), 3181–3184. https://doi.org/10.1111/gcb.15062

Roy, H. E., Rabitsch, W., Scalera, R., Stewart, A., Gallardo, B., Genovesi, P., Essl, F., Adriaens, T., Bacher, S., Booy, O., Branquart, E., Brunel, S., Copp, G. H., Dean, H., D'hondt, B., Josefsson, M., Kenis, M., Kettunen, M., Linnamagi, M., … Zenetos, A. (2018). Developing a framework of minimum standards for the risk assessment of alien species. *Journal of Applied Ecology*, *55*(2), 526–538. https://doi.org/10.1111/1365-2664.13025

Roy, S., Smith, G. C., & Russell, J. C. (2009). The eradication of invasive mammal species: Can adaptive resource management fill the gaps in our knowledge? *Human–Wildlife Conflicts*, *3*(35), 30–40.

Royimani, L., Mutanga, O., Odindi, J., Dube, T., & Matongera, T. N. (2019). Advancements in satellite remote sensing for mapping and monitoring of alien invasive plant species (AIPs). *Physics and Chemistry of the Earth, Parts A/B/C*, *112*, 237–245. https://doi.org/10.1016/j.pce.2018.12.004

RSPCA. (2019). What is rabbit calicivirus and how do I protect my rabbit from rabbit haemorrhagic disease? – RSPCA Knowledgebase [Royal Society for the Prevention of Cruelty to Animals, Australia]. *RSPCA Knowledgebase*. https://kb.rspca.org.au/knowledge-base/what-is-rabbit-calicivirus-and-how-do-i-protect-my-rabbit-from-rabbit-haemorrhagic-disease/

RSPCA. (2021). *About Us*. Royal Society for the Prevention of Cruelty to Animals, Australia. https://www.rspca.org.au/about-us

Ruhi, A., Catford, J. A., Cross, W. F., Escoriza, D., & Olden, J. D. (2019). Chapter 3—Understanding the Nexus Between Hydrological Alteration And Biological Invasions. In S. Sabater, A. Elosegi, & R. Ludwig (Eds.), *Multiple Stressors in River Ecosystems: Status, Impacts and Prospects for the Future* (pp. 45–64). Elsevier. https://doi.org/10.1016/B978-0-12-811713-2.00003-0

Rundel, P. W., Graham, E. A., Allen, M. F., Fisher, J. C., & Harmon, T. C. (2009). Environmental sensor networks in ecological research. *New Phytologist*, *182*(3), 589–607. https://doi.org/10.1111/j.1469-8137.2009.02811.x

Russell, J. C., Innes, J. G., Brown, P. H., & Byrom, A. E. (2015). Predator-Free New Zealand: Conservation Country. *BioScience*, *65*(5), 520–525. https://doi.org/10.1093/biosci/biv012

Rwomushana, I., Bateman, M., Beale, T., Beseh, P., Cameron, K., Chiluba, M., Clottey, V., Davis, T., Day, R., Early, R., Godwin, J., Gonzalez-Moreno, P., Kansiime, M., Kenis, M., Makale, F., Mugambi, I., Murphy, S., Nunda, W., Phiri, N., … Tambo, J. (2018). *Fall armyworm: Impacts and implications for Africa. Evidence Note Update, October 2018*. CABI. https://www.invasive-species.org/wp-content/uploads/sites/2/2019/02/FAW-Evidence-Note-October-2018.pdf

Salafsky, N., Margoluis, M., & Redford, K. H. (2001). *Adaptive management: A tool for conservation practitioners*. Biodiversity Support Program. http://hdl.handle.net/10919/67624

Salafsky, N., Salzer, D., Stattersfield, A. J., Hilton-Taylor, C., Neugarten, R., Butchart, S. H. M., Collen, B., Cox, N., Master, L. L., O'connor, S., & Wilkie, D. (2008). A Standard Lexicon for Biodiversity Conservation: Unified Classifications of Threats and Actions. *Conservation Biology*, *22*(4), 897–911. https://doi.org/10.1111/j.1523-1739.2008.00937.x

Sammarco, P. W., Porter, S. A., Sinclair, J., & Genazzio, M. (2013). Depth distribution of a new invasive coral (Gulf of Mexico) – *Tubastraea micranthus*, comparisons with *T. coccinea*, and implications for control. *Management of Biological Invasions*, *4*(4), 291–303. https://doi.org/10.3391/mbi.2013.4.4.04

Samways, M. J., Hitchins, P. M., Bourquin, O., & Henwood, J. (2010). Restoration of a tropical island: Cousine Island, Seychelles. *Biodiversity and Conservation*, *19*(2), 425–434. https://doi.org/10.1007/s10531-008-9524-z

Sandvik, H., Hilmo, O., Finstad, A. G., Hegre, H., Moen, T. L., Rafoss, T., Skarpaas, O., Elven, R., Sandmark, H., & Gederaas, L. (2019). Generic ecological impact assessment of alien species (GEIAA): The third generation of assessments in Norway. *Biological Invasions*, *21*(9), 2803–2810. https://doi.org/10.1007/s10530-019-02033-6

Sankaran, K. V. (2008). *Giant African Snail. Pest Fact Sheet. Asia-Pacific Forest Invasive Species Network, FAO, Bangkok, apfisn.net*. https://apfisn.net/wp-content/uploads/2018/07/Achatina-fulica_0.pdf

Sankaran, K. V., & Suresh, T. A. (2013). *Evaluation of Classical Biological Control of* Mikania micrantha *with* Puccinia spegazzinii (Final Report of Research Project No.416/2003 KFRI Research report No.472/2013). Kerala Forest Research Institute.

Santagata, S., Gasiūnaite, Z. R., Verling, E., Cordell, J. R., Eason, K., Cohen, J. S., Bacela, K., Quilez-Badia, G., Johengen, T. H., Reid, D. F., & Ruiz, G. M. (2008). Effect of osmotic shock as a management strategy to reduce transfers of non-indigenous species among low-salinity ports by ships. *Aquatic Invasions*, *3*(1), 61–76. https://doi.org/10.3391/ai.2008.3.1.10

Saul, W.-C., Roy, H. E., Booy, O., Carnevali, L., Chen, H.-J., Genovesi, P., Harrower, C. A., Hulme, P. E., Pagad, S., Pergl, J., & Jeschke, J. M. (2017). Assessing patterns in introduction pathways of alien species by linking major invasion data bases. *Journal of Applied Ecology*, *54*(2), 657–669. https://doi.org/10.1111/1365-2664.12819

Saunders, G., Cooke, B., McColl, K., Shine, R., & Peacock, T. (2010). Modern approaches for the biological control of vertebrate pests: An Australian perspective. *Biological Control*, *52*(3), 288–295. https://doi.org/10.1016/j.biocontrol.2009.06.014

Sax, D. F., Gaines, S. D., & Brown, J. H. (2002). Species Invasions Exceed Extinctions on Islands Worldwide: A Comparative Study of Plants and Birds. *The American Naturalist*, *160*(6), 766–783. https://doi.org/10.1086/343877

Scapin, P., Ulbano, M., Ruggiero, C., Balduzzi, A., Marsan, A., Ferrari, N., & Bertolino, S. (2019). Surgical sterilization of male and female grey squirrels (*Sciurus carolinensis*) of an urban population introduced in Italy. *Journal of Veterinary Medical Science*, *advpub*. https://doi.org/10.1292/jvms.18-0319

Scasta, J. D., Engle, D. M., Fuhlendorf, S. D., Redfearn, D. D., & Bidwell, T. G. (2015). Meta-Analysis of Exotic Forages as Invasive Plants in Complex Multi-Functioning Landscapes. *Invasive Plant Science and Management*, *8*(3), 292–306. https://doi.org/10.1614/IPSM-D-14-00076.1

Schirrmeier, H., Reimann, I., Köllner, B., & Granzow, H. (1999). Pathogenic, antigenic and molecular properties of rabbit haemorrhagic disease virus (RHDV) isolated from vaccinated rabbits: Detection and characterization of antigenic variants. *Archives of Virology*, *144*(4), 719–735. https://doi.org/10.1007/s007050050538

Schlipalius, D. I., Ebert, P. R., & Collins, P. J. (2015). An emerging international picture of phosphine resistance: Opportunities for global cooperation. *Session 1: Emerging Global Issues in Stored Product Protection*, 10–14. https://doi.org/10.14455/DOA.res.2014.3

Schmidt-Lebuhn, A. N., & Norton, G. (2017). An Online Multi-Access Identification Key to the Propagules of Selected Biosecurity-Relevant Asteraceae (Daisy or Sunflower Family). *Invasive Plant Science and Management*, *10*(2), 210–211. https://doi.org/10.1017/inp.2017.17

Schnase, J. L., Smith, J. A., Stohlgren, T. J., Graves, S., & Trees, C. (2002). Biological invasions: A challenge in ecological forecasting. *IEEE International Geoscience and Remote Sensing Symposium*, *1*, 122–124 vol.1. https://doi.org/10.1109/IGARSS.2002.1024961

Schneider, K., Camac, J., Dodd, A., Fraser, H., Gomboso, J., Gibert, A., Kompas, T., Lane, S., Robinson, A., & Spring, D. (2020). *Evaluating the health of Australia's biosecurity system*. https://cebra.unimelb.edu.au/__data/assets/pdf_file/0011/3423278/Endorsed-CEBRA-170714-Final-Report-June2020.pdf

Schrader, G., & Unger, J.-G. (2003). Plant Quarantine as a Measure Against Invasive Alien Species: The Framework of the International Plant Protection Convention and the plant health regulations in the European Union. *Biological Invasions*, *5*(4), 357–364. https://doi.org/10.1023/B:BINV.0000005567.58234.b9

Schwarzländer, M., Hinz, H. L., Winston, R. L., & Day, M. D. (2018). Biological control of weeds: An analysis of introductions, rates of establishment and estimates of success, worldwide. *BioControl*, *63*(3), 319–331. https://doi.org/10.1007/s10526-018-9890-8

Scianni, C., & Georgiades, E. (2019). Vessel In-Water Cleaning or Treatment: Identification of Environmental Risks and Science Needs for Evidence-Based Decision Making. *Frontiers in Marine Science*, *6*, 467. https://doi.org/10.3389/fmars.2019.00467

Scott, J. K. (1995). 7 Classical biological control of plant pathogens. In J. H. Andrews & I. C. Tommerup (Eds.), *Advances in Plant Pathology* (Vol. 11, pp. 131–146). Academic Press. https://doi.org/10.1016/S0736-4539(06)80009-7

Scott, J. K., McKirdy, S. J., van der Merwe, J., Green, R., Burbidge, A. A., Pickles, G., Hardie, D. C., Morris, K., Kendrick, P. G., Thomas, M. L., Horton, K. L., O'Connor, S. M., Downs, J., Stoklosa, R., Lagdon, R., Marks, B., Nairn, M., & Mengersen, K. (2017). Zero-tolerance biosecurity protects high-conservation-value island nature reserve. *Scientific Reports*, *7*(1), 772. https://doi.org/10.1038/s41598-017-00450-y

Scott, J. K., & Panetta, F. D. (1993). Predicting the Australian Weed Status of Southern African Plants. *Journal of Biogeography*, *20*(1), 87–93. https://doi.org/10.2307/2845742

Secord, D. (2003). Biological control of marine invasive species: Cautionary tales and land-based lessons. *Biological Invasions*, *5*(1/2), 117–131. https://doi.org/10.1023/A:1024054909052

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., … Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications*, *8*(1), 14435. https://doi.org/10.1038/ncomms14435

Senterre, B., & Dine, M. L. P. (2022). *Toward a Seychelles Invasive Species Strategy and Action Plan (ISSAP): Biological factors* [Final Report]. Unpublished. https://doi.org/10.13140/RG.2.2.18218.70081

Sergeant, ESG. (2018). *Epitools Epidemiological Calculators*. Ausvet. http://epitools.ausvet.com.au.

Seto, K. C., Parnell, S., & Elmqvist, T. (2013). A Global Outlook on Urbanization. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, S. Parnell, M. Schewenius, M. Sendstad, K. C. Seto, & C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities* (pp. 1–12). Springer Open. https://doi.org/10.1007/978-94-007-7088-1_1

Seto, K. C., & Shepherd, J. M. (2009). Global Urban Land-Use Trends and Climate Impacts. *Current Opinion in Environmental Sustainability*, *1*(1), 89–95. https://doi.org/10.1016/j.cosust.2009.07.012

Setterfield, S. A., Douglas, M. M., Petty, A. M., Bayliss, P., Ferdinands, K. B., & Winderlich, S. (2013). Invasive Plants in the Floodplains of Australia's Kakadu National Park. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas. Patterns, Problems and Challenges.* (Vol. 7, pp. 167–189). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_9

Setyawati, T., Sunardi, Tjitrosoedirdjo, S., T. Pullaiah, & Ielmini, M. R. (2021). Invasive Alien Plant Species Management in Indonesia. In *Invasive Alien Species: Observations and Issues from Around the World* (pp. 73–102). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781119607045.ch16

Shackleton, R. T., Adriaens, T., Brundu, G., Dehnen-Schmutz, K., Estévez, R. A., Fried, J., Larson, B. M. H., Liu, S., Marchante, E., Marchante, H., Moshobane, M. C., Novoa, A., Reed, M., & Richardson, D. M. (2019). Stakeholder engagement in the study and management of invasive alien species. *Journal of Environmental Management*, *229*, 88–101. https://doi.org/10.1016/j.jenvman.2018.04.044

Shackleton, R. T., Bertzky, B., Wood, L. E., Bunbury, N., Jäger, H., van Merm, R., Sevilla, C., Smith, K., Wilson, J. R. U., Witt, A. B. R., & Richardson, D. M. (2020). Biological invasions in World Heritage Sites: Current status and a proposed monitoring and reporting framework. *Biodiversity and Conservation*, *29*(11), 3327–3347. https://doi.org/10.1007/s10531-020-02026-1

Shackleton, R. T., Foxcroft, L. C., Pyšek, P., Wood, L. E., & Richardson, D. M. (2020). Assessing biological invasions in protected areas after 30 years: Revisiting nature reserves targeted by the 1980s SCOPE programme. *Biological Conservation*, *243*, 108424. https://doi.org/10.1016/j.biocon.2020.108424

Shackleton, R. T., Larson, B. M. H., Novoa, A., Richardson, D. M., & Kull, C. A. (2019). The human and social dimensions of invasion science and management. *Journal of Environmental Management*, *229*, 1–9. https://doi.org/10.1016/j.jenvman.2018.08.041

Shackleton, R. T., Le Maitre, D. C., Pasiecznik, N. M., & Richardson, D. M. (2014). *Prosopis*: A global assessment of the biogeography, benefits, impacts and management of one of the world's worst woody invasive plant taxa. *AoB PLANTS*, *6*, plu027. https://doi.org/10.1093/aobpla/plu027

Shackleton, R. T., Witt, A. B. R., Piroris, F. M., & van Wilgen, B. W. (2017). Distribution and socio-ecological impacts of the invasive alien cactus *Opuntia stricta* in eastern Africa. *Biological Invasions*, *19*(8), 2427–2441. https://doi.org/10.1007/s10530-017-1453-x

Shackleton, S., Kirby, D., & Gambiza, J. (2011). Invasive plants – friends or foes? Contribution of prickly pear (*Opuntia ficus-indica*) to livelihoods in Makana Municipality, Eastern Cape, South Africa. *Development Southern Africa*, *28*(2), 177–193. https://doi.org/10.1080/0376835X.2011.570065

Shanmuganathan, T., Pallister, J., Doody, S., McCallum, H., Robinson, T., Sheppard, A., Hardy, C., Halliday, D., Venables, D., Voysey, R., Strive, T., Hinds, L., & Hyatt, A. (2010). Biological control of the cane toad in Australia: A review. *Animal Conservation*, *13*(s1), 16–23. https://doi.org/10.1111/j.1469-1795.2009.00319.x

Shao, H., Xi, N., & Zhang, Y. (2018). Microemulsion formulation of a new biopesticide to control the diamondback moth (Lepidoptera: Plutellidae). *Scientific Reports*, *8*(1), 10565. https://doi.org/10.1038/s41598-018-28626-0

Sharma, A., Shukla, A., Attri, K., Kumar, M., Kumar, P., Suttee, A., Singh, G., Barnwal, R. P., & Singla, N. (2020). Global trends in pesticides: A looming threat and viable alternatives. *Ecotoxicology and Environmental Safety*, *201*, 110812. https://doi.org/10.1016/j.ecoenv.2020.110812

Sharov, A. A., Leonard, D., Liebhold, A. M., Roberts, E. A., & Dickerson, W. (2002). "Slow The Spread": A National Program to Contain the Gypsy Moth. *Journal of Forestry*, *100*(5), 30–36. https://doi.org/10.1093/jof/100.5.30

Sharov, A. A., & Liebhold, A. M. (1998). Model of slowing the spread of gypsy moth (Lepidoptera: Lymantriidae) with a barrier zone. *Ecological Applications*, *8*(4), 1170–1179. https://doi.org/10.2307/2640970

Sharp, T., & Saunders, G. (2011). *A model for assessing the relative humaneness of pest animal control methods* (Second). Australian Government Department of Agriculture, Fisheries and Forestry, Canberra, ACT.

Shaw, A., Miller, K. K., & Wescott, G. (2017). Australian native gardens: Is there scope for a community shift? *Landscape and Urban Planning*, *157*, 322–330. https://doi.org/10.1016/j.landurbplan.2016.07.009

Shea, K., Possingham, H. P., Murdoch, W. W., & Roush, R. (2002). Active Adaptive Management in Insect Pest and Weed Control: Intervention with a Plan for Learning. *Ecological Applications*, *12*(3), 927–936. https://doi.org/10.1890/1051-0761(2002)012[0927:AAMIIP]2.0.CO;2

Sheail, J. (2003). Government and the Management of an Alien Pest Species: A British perspective. *Landscape Research*, *28*(1), 101–111. https://doi.org/10.1080/01426390306528

Sheley, R. L., Svejcar, T. J., & Maxwell, B. D. (1996). A Theoretical Framework for Developing Successional Weed Management Strategies on Rangeland. *Weed Technology*, *10*(4), 766–773. https://doi.org/10.1017/S0890037X00040793

Shephard, R. W., Williams, S. H., & Beckett, S. D. (2016). Farm economic impacts of bovine Johne's disease in endemically infected Australian dairy herds. *Australian Veterinary Journal*, *94*(7), 232–239. https://doi.org/10.1111/avj.12455

Sheppard, A. W., & Glanzing, A. (2021). *Fighting plagues and predators Australia's path towards a pest and weed-free future*. CSIRO. https://www.csiro.au/-/media/News-releases/2021/Fighting-plagues-and-predators/Final-FightingPlaguesPredators_WEB_211126.pdf

Sheppard, A. W., Hill, R., DeClerck-Floate, R. A., McClay, A., Olckers, T., Quimby, P. C., & Zimmermann, H. G. (2003). A global review of risk-benefit-cost analysis for the introduction of classical biological control agents against weeds: A crisis in the making? *Biocontrol News and Information*, *24*(4), 91N-108N. https://www.cabdirect.org/cabdirect/abstract/20043008816

Sheppard, A. W., Hodge, P., Paynter, Q., & Rees, M. (2002). Factors Affecting Invasion and Persistence of Broom Cytisus scoparius in Australia. *Journal of Applied Ecology*, *39*(5), 721–734. JSTOR. https://www.jstor.org/stable/827200

Sheppard, A. W., Paynter, Q., Mason, P., Murphy, S., Stoett, P., Cowan, P., Brodeur, J., Warner, K., Villegas, C., Shaw, R., Hinz, H., Hill, M., & Genovesi, P. (2019). IUCN SSC Invasive Species Specialist Group. The Application of Biological Control for the Management of Established Invasive Alien Species Causing Environmental Impacts. *The Application of Biological Control for the Management of Established Invasive Alien Species Causing Environmental Impacts. The Secretariat of the Convention on Biological Diversity Technical Series*, *Technical Series No. 91. Montreal, Canada 88 pages.*(91). https://www.cbd.int/doc/publications/cbd-ts-91-en.pdf

Shine, C. (2005). Overview of the management of invasive alien species from the environmental perspective. In*: Identification of risks and management of invasive alien species using the IPPC framework. Proceedings of the workshop on invasive alien species and the International Plant Protection Convention (IPPC), IPPC Secretariat, Braunschweig, Germany, 22-26 September 2003. Rome, Italy, FAO* (p. xii + 301). https://assets.ippc.int/static/media/files/publications/en/1065703408882_FRANCE_IUCN_Clare_Shine_1.pdf

Shrestha, B. B., Shrestha, U. B., Sharma, K. P., Thapa-Parajuli, R. B., Devkota, A., & Siwakoti, M. (2019). Community perception and prioritization of invasive alien plants in Chitwan-Annapurna Landscape, Nepal. *Journal of Environmental Management*, *229*, 38–47. https://doi.org/10.1016/j.jenvman.2018.06.034

Simberloff, D. (2001). Eradication of island invasives: Practical actions and results achieved. *Trends in Ecology & Evolution*, *16*(6), 273–274. https://doi.org/10.1016/S0169-5347(01)02154-1

Simberloff, D. (2003). Eradication-preventing invasions at the outset. *Weed Science*, *51*(2), 247–253. https://doi.org/10.1614/0043-1745(2003)051[0247:EPIATO]2.0.CO;2

Simberloff, D. (2006). Risk Assessments, Blacklists, and White Lists for Introduced Species: Are Predictions Good Enough to Be Useful? *Agricultural and Resource Economics Review*, *35*(1), 1–10. https://doi.org/10.1017/S1068280500010005

Simberloff, D. (2009). We can eliminate invasions or live with them. Successful management projects. *Biological Invasions*, *11*(1), 149–157. https://doi.org/10.1007/s10530-008-9317-z

Simberloff, D. (2013). Eradication: Pipe Dream or Real Option? In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant Invasions in Protected Areas* (Vol. 7, pp. 549–559). Springer Netherlands. https://doi.org/10.1007/978-94-007-7750-7_25

Simberloff, D. (2021). Maintenance management and eradication of established aquatic invaders. *Hydrobiologia*, *848*(9), 2399–2420. https://doi.org/10.1007/s10750-020-04352-5

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vilà, M. (2013). Impacts of biological invasions: What's what and the way forward. *Trends in Ecology & Evolution*, *28*(1), 58–66. https://doi.org/10.1016/j.tree.2012.07.013

Simberloff, D., Parker, I. M., & Windle, P. N. (2005). Introduced species policy, management, and future research needs. *Frontiers in Ecology and the Environment*, *3*(1), 12–20. https://doi.org/10.1890/1540-9295(2005)003[0012:ISPMAF]2.0.CO;2

Simberloff, D., & Stiling, P. (1996). How risky is biological control? *Ecology*, *77*(7), 1965–1974. https://doi.org/10.2307/2265693

Simon, S., Otto, M., & Engelhard, M. (2018). Synthetic gene drive: Between continuity and novelty. *EMBO Reports*, *19*(5), e45760. https://doi.org/10.15252/embr.201845760

Simpson, A. (2004). The Global Invasive Species Information Network: What's in It for You? *BioScience*, *54*(7), 613–614. https://doi.org/10.1641/0006-3568(2004)054[0613:TGISIN]2.0.CO;2

Simpson, A., Jarnevich, C., Madsen, J., Westbrooks, R., Fournier, C., Mehrhoff, L., Browne, M., Graham, J., & Sellers, E. (2009). Invasive species information networks: Collaboration at multiple scales for prevention, early detection, and rapid response to invasive alien species. *Biodiversity*, *10*(2–3), 5–13. https://doi.org/10.1080/14888386.2009.9712839

Simpson, A., Sellers, E., Grosse, A., & Xie, Y. (2006). Essential elements of online information networks on invasive alien species. *Biological Invasions*, *8*(7), 1579–1587. https://doi.org/10.1007/s10530-005-5850-1

Sims, B. G., Thierfelder, C., Kienzle, J., Friedrich, T., & Kassam, A. (2012). Development of the Conservation Agriculture Equipment Industry in Sub-Saharan Africa. *Applied Engineering in Agriculture*, *28*(6), 813–823. https://doi.org/10.13031/2013.42472

Sinclair, K., Curtis, A. L., Hacker, R. B., & Atkinson, T. (2020). Stakeholder judgements of the social acceptability of control practices for kangaroos, unmanaged goats and feral pigs in the south-eastern rangelands of Australia. *The Rangeland Journal*, *41*(6), 485–496. https://doi.org/10.1071/RJ19047

Smith, D., & Dunbabin, M. (2007). Automated Counting of the Northern Pacific Sea Star in the Derwent Using Shape Recognition. *9th Biennial Conference of the Australian Pattern Recognition Society on Digital Image Computing Techniques and Applications (DICTA 2007)*, 500–507. https://doi.org/10.1109/DICTA.2007.4426838

Smith, K., & Kraaij, T. (2020). Research note: Trail runners as agents of alien plant introduction into protected areas. *Journal of Outdoor Recreation and Tourism*, *31*, 100315. https://doi.org/10.1016/j.jort.2020.100315

Smith, P. J., Page, M., Handley, S. J., McVeagh, S. M., & Ekins, M. (2007). First record of the Australian ascidian *Eudistoma elongatum* in northern New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *41*(4), 347–355. https://doi.org/10.1080/00288330709509924

Smith, P. J., Webber, W. R., McVeagh, S. M., Inglis, G. J., & Gust, N. (2003). DNA and morphological identification of an invasive swimming crab, *Charybdis japonica*, in New Zealand waters. *New Zealand Journal of Marine and Freshwater Research*, *37*(4), 753–762. https://doi.org/10.1080/00288330.2003.9517205

Smith, R. G., Maxwell, B. D., Menalled, F. D., & Rew, L. J. (2006). Lessons from agriculture may improve the management of invasive plants in wildland systems. *Frontiers in Ecology and the Environment*, *4*(8), 428–434. https://doi.org/10.1890/1540-9295(2006)4[428:LFAMIT]2.0.CO;2

Solow, A., Seymour, A., Beet, A., & Harris, S. (2008). The untamed shrew: On the termination of an eradication programme for an introduced species. *Journal of Applied Ecology*, *45*(2), 424–427. https://doi.org/10.1111/j.1365-2664.2007.01446.x

Sommer, P., Kusy, B., Jurdak, R., Kottege, N., Liu, J., Zhao, K., McKeown, A., & Westcott, D. (2016). From the lab into the wild: Design and deployment methods for multi-modal tracking platforms. *Pervasive and Mobile Computing*, *30*, 1–17. https://doi.org/10.1016/j.pmcj.2015.09.003

Sor, R., Ngor, P. B., Boets, P., Goethals, P. L. M., Lek, S., Hogan, Z. S., & Park, Y.-S. (2020). Patterns of Mekong Mollusc Biodiversity: Identification of Emerging Threats and Importance to Management and Livelihoods in a Region of Globally Significant Biodiversity and Endemism. *Water*, *12*(9), Article 9. https://doi.org/10.3390/w12092619

Sosa, A. J., Jiménez, N. L., Faltlhauser, A. C., Righetti, T., Mc Kay, F., Bruzzone, O. A., Stiers, I., & Fernández Souto, A. (2021). The educational community and its knowledge and perceptions of native and invasive alien species. *Scientific Reports*, *11*(1), Article 1. https://doi.org/10.1038/s41598-021-00683-y

Soubeyran, Y., Meyer, J.-Y., Lebouvier, M., De Thoisy, B., Lavergne, C., Urtizberea, F., & Kirchner, F. (2015). Dealing with invasive alien species in the French overseas territories: Results and benefits of a 7-year Initiative. *Biological Invasions*, *17*(2), 545–554. https://doi.org/10.1007/s10530-014-0766-2

Soukhaphon, A., Baird, I. G., & Hogan, Z. S. (2021). The Impacts of Hydropower Dams in the Mekong River Basin: A Review. *Water*, *13*(3), 265. https://doi.org/10.3390/w13030265

Sparks, T. C., & Nauen, R. (2015). IRAC: Mode of action classification and insecticide resistance management. *Pesticide Biochemistry and Physiology*, *121*, 122–128. https://doi.org/10.1016/j.pestbp.2014.11.014

Spatz, D. R., Holmes, N. D., Will, D. J., Hein, S., Carter, Z. T., Fewster, R. M., Keitt, B., Genovesi, P., Samaniego, A., Croll, D. A., Tershy, B. R., & Russell, J. C. (2022). The global contribution of invasive vertebrate eradication as a key island restoration tool. *Scientific Reports*, *12*(1), Article 1. https://doi.org/10.1038/s41598-022-14982-5

Spear, D., Foxcroft, L. C., Bezuidenhout, H., & McGeoch, M. A. (2013). Human population density explains alien species richness in protected areas. *Biological Conservation*, *159*, 137–147. https://doi.org/10.1016/j.biocon.2012.11.022

St George, T. D. (1985). Studies on the pathogenesis of bovine ephemeral fever in sentinel cattle. I. Virology and serology. *Veterinary Microbiology*, *10*(6), 493–504. https://doi.org/10.1016/0378-1135(85)90058-6

Stanley, M. (2004). Review of the efficacy of baits used for ant control and eradication. *Landcare Research Contract Report: LC0405/044*, 74. https://littlefireants.com/wp-content/uploads/Stanley-2004_compressed.pdf

Staples, E. J., & Viswanathan, S. (2008). Detection of Contrabands in Cargo Containers Using a High-Speed Gas Chromatograph with Surface Acoustic Wave Sensor. *Industrial & Engineering Chemistry Research*, *47*(21), 8361–8367. https://doi.org/10.1021/ie701703y

Stephens, A. E. A., Krannitz, P. G., & Myers, J. H. (2009). Plant community changes after the reduction of an invasive rangeland weed, diffuse knapweed, *Centaurea diffusa*. *Biological Control*, *51*(1), 140–146. https://doi.org/10.1016/j.biocontrol.2009.06.015

Stevenson, M. (2004). *Hunters and Bureaucrats: Power, Knowledge and Aboriginal-State Relations in the Southwest Yukon*. https://journalhosting.ucalgary.ca/index.php/arctic/article/download/63610/47546

Stilgoe, J., Owen, R., & Macnaghten, P. (2020). Developing a framework for responsible innovation. *Research Policy*, *42*(9), 1568–1580. https://doi.org/10.1016/j.respol.2013.05.008

Stohlgren, T. J., & Jarnevich, C. S. (2009). Risk assessment of invasive species. In M. N. Clout & P. A. Williams (Eds.), *Invasive species management. A handbook of principles and techniques.* (pp. 19–35). Oxford University Press. https://global.oup.com/academic/product/invasive-species-management-9780199216338?cc=jp&lang=en&

Stokes, K. E., O°Neill, K. P., Montgomery, W. I., Dick, J. T. A., Maggs, C. A., & McDonald, R. A. (2006). The Importance of Stakeholder Engagement in Invasive Species Management: A Cross-jurisdictional Perspective in Ireland. *Biodiversity and Conservation*, 15(8), 2829–2852. https://doi.org/10.1007/s10531-005-3137-6

Stokols, D. (1996). Translating Social Ecological Theory into Guidelines for Community Health Promotion. *American Journal of Health Promotion*, *10*(4), 282–298. https://doi.org/10.4278/0890-1171-10.4.282

Storm, D. J., Samuel, M. D., Van Deelen, T. R., Malcolm, K. D., Rolley, R. E., Frost, N. A., Bates, D. P., & Richards, B. J. (2011). Comparison of visual-based helicopter and fixed-wing forward-looking infrared surveys for counting white-tailed deer *Odocoileus virginianus*. *Wildlife Biology*, *17*(4), 431–440. https://doi.org/10.2981/10-062

Strand, J. F. (2000). Some agrometeorological aspects of pest and disease management for the 21st century. *Agricultural and Forest Meteorology*, *103*(1), 73–82. https://doi.org/10.1016/S0168-1923(00)00119-2

Stringham, O. C., García-Díaz, P., Toomes, A., Mitchell, L., Ross, J. V., & Cassey, P. (2021). Live reptile smuggling is predicted by trends in the legal exotic pet trade. *Conservation Letters*, *14*(6), e12833. https://doi.org/10.1111/conl.12833

Suarez-Menendez, M., Planes, S., Garcia-Vazquez, E., & Ardura, A. (2020). Early Alert of Biological Risk in a Coastal Lagoon Through eDNA Metabarcoding. *Frontiers in Ecology and Evolution*, *8*, 9. https://doi.org/10.3389/fevo.2020.00009

Suckling, D. M. (2015). Can we replace toxicants, achieve biosecurity, and generate market position with semiochemicals? *Frontiers in Ecology and Evolution*, *3*. https://doi.org/10.3389/fevo.2015.00017

Suckling, D. M., Stringer, L. D., Stephens, A. E., Woods, B., Williams, D. G., Baker, G., & El-Sayed, A. M. (2014). From integrated pest management to integrated pest eradication: Technologies and future needs. *Pest Management Science*, *70*(2), 179–189. https://doi.org/10.1002/ps.3670

Sun, J., Lu, M., Gillette, N. E., & Wingfield, M. J. (2013). Red Turpentine Beetle: Innocuous Native Becomes Invasive Tree Killer in China. *Annual Review of Entomology*, *58*(1), 293–311. https://doi.org/10.1146/annurev-ento-120811-153624

Sun, Y., Ding, J., Siemann, E., & Keller, S. R. (2020). Biocontrol of invasive weeds under climate change: Progress, challenges and management implications. *Current Opinion in Insect Science*, *38*, 72–78. https://doi.org/10.1016/j.cois.2020.02.003

Sundaram, B., Krishnan, S., Hiremath, A. J., & Joseph, G. (2012). Ecology and Impacts of the Invasive Species, *Lantana camara*, in a Social-Ecological System in South India: Perspectives from Local Knowledge. *Human Ecology*, *40*(6), 931–942. https://doi.org/10.1007/s10745-012-9532-1

Sutherland, W. J. (Ed.). (2022). *Transforming Conservation: A Practical Guide to Evidence and Decision Making*. Open Book Publishers. https://doi.org/10.11647/obp.0321

Sutherland, W. J., Taylor, N. G., MacFarlane, D., Amano, T., Christie, A. P., Dicks, L. V., Lemasson, A. J., Littlewood, N. A., Martin, P. A., Ockendon, N., Petrovan, S. O., Robertson, R. J., Rocha, R., Shackelford, G. E., Smith, R. K., Tyler, E. H. M., & Wordley, C. F. R. (2019). Building a tool to overcome barriers in research-implementation spaces: The Conservation Evidence database. *Biological Conservation*, *238*, 108199. https://doi.org/10.1016/j.biocon.2019.108199

Sutherst, R. W., Constable, F., Finlay, K. J., Harrington, R., Luck, J., & Zalucki, M. P. (2011). Adapting to crop pest and pathogen risks under a changing climate. *WIREs Climate Change*, *2*(2), 220–237. https://doi.org/10.1002/wcc.102

Swathy, N. S. (2021). *Early detection and Rapid Response (EDRR) Strategy* *and Genome Sequencing for Controlling* Ceroplastes cirripediformis *with reference to its biological studies* (M.Sc. thesis in Zoology).

Sweeney, J., De Groot, P., MacDonald, L., Smith, S., Cocquempot, C., Kenis, M., & Gutowski, J. M. (2004). Host Volatile Attractants and Traps for Detection of *Tetropium fuscum* (F.), *Tetropium castaneum* L., and Other Longhorned Beetles (Coleoptera: Cerambycidae). *Environmental Entomology*, *33*(4), 844–854. https://doi.org/10.1603/0046-225X-33.4.844

Taitingfong, R. I. (2019). Islands as Laboratories: Indigenous Knowledge and Gene Drives in the Pacific. *Human Biology*, *91*(3), 179–188. https://doi.org/10.13110/humanbiology.91.3.01

Tambo, J. A., Aliamo, C., Davis, T., Mugambi, I., Romney, D., Onyango, D. O., Kansiime, M., Alokit, C., & Byantwale, S. T. (2019). The impact of ICT-enabled extension campaign on farmers' knowledge and management of fall armyworm in Uganda. *PLoS ONE*, *14*(8), e0220844. https://doi.org/10.1371/journal.pone.0220844

Tambo, J. A., Day, R. K., Lamontagne-Godwin, J., Silvestri, S., Beseh, P. K., Oppong-Mensah, B., Phiri, N. A., & Matimelo, M. (2020). Tackling fall armyworm (*Spodoptera frugiperda*) outbreak in Africa: An analysis of farmers' control actions. *International Journal of Pest Management*, *66*(4), 298–310. https://doi.org/10.1080/09670874.2019.1646942

Tambo, J. A., Kansiime, M. K., Mugambi, I., Rwomushana, I., Kenis, M., Day, R. K., & Lamontagne-Godwin, J. (2020). Understanding smallholders' responses to fall armyworm (*Spodoptera frugiperda*) invasion: Evidence from five African countries. *Science of the Total Environment*, *740*, 140015. https://doi.org/10.1016/j.scitotenv.2020.140015

Tambo, J. A., Romney, D., Mugambi, I., Mbugua, F., Bundi, M., Uzayisenga, B., Matimelo, M., & Ndhlovu, M. (2021). Can plant clinics enhance judicious use of pesticides? Evidence from Rwanda and Zambia. *Food Policy*, *101*, 102073. https://doi.org/10.1016/j.foodpol.2021.102073

Tamburri, M. N., Davidson, I. C., First, M. R., Scianni, C., Newcomer, K., Inglis, G. J., Georgiades, E. T., Barnes, J. M., & Ruiz, G. M. (2020). In-Water Cleaning and Capture to Remove Ship Biofouling: An Initial Evaluation of Efficacy and Environmental Safety. *Frontiers in Marine Science*, *7*, 437. https://doi.org/10.3389/fmars.2020.00437

Tana, T. (2014). The MPI animal general surveillance programme. *Surveillance*, *41*(2), 5–8.

Tanentzap, A. J., Bazely, D. R., Williams, P. A., & Hoogensen, G. (2009). A Human Security Framework for the Management of Invasive Nonindigenous Plants. *Invasive Plant Science and Management*, *2*(2), 99–109. https://doi.org/10.1614/IPSM-08-127.1

Tay, W. T., Rane, R. V., Padovan, A., Walsh, T. K., Elfekih, S., Downes, S., Nam, K., d'Alençon, E., Zhang, J., Wu, Y., Nègre, N., Kunz, D., Kriticos, D. J., Czepak, C., Otim, M. H., & Gordon, K. H. J. (2022). Global population genomic signature of *Spodoptera frugiperda* (fall armyworm) supports complex introduction events across the Old World. *Communications Biology*, *5*, 297. https://doi.org/10.1038/s42003-022-03230-1

Teem, J. L., Alphey, L., Descamps, S., Edgington, M. P., Edwards, O., Gemmell, N., Harvey-Samuel, T., Melnick, R. L., Oh, K. P., Piaggio, A. J., Saah, J. R., Schill, D., Thomas, P., Smith, T., & Roberts, A. (2020). Genetic Biocontrol for Invasive Species. *Frontiers in Bioengineering and Biotechnology*, *8*, 452. https://doi.org/10.3389/fbioe.2020.00452

Tenllado, F., & Díaz-Ruíz, J. R. (2001). Double-Stranded RNA-Mediated Interference with Plant Virus Infection. *Journal of Virology*, *75*(24), 12288–12297. https://doi.org/10.1128/JVI.75.24.12288-12297.2001

Terefe, B., Williams, F., & Lamontagne-Godwin, J. (2020). *Invasive species management—Integrating a gender perspective*. CABI. https://doi.org/10.1079/CABICOMM-62-8140

The Commonwealth of Australia. (2016). *Wetlands and Indigenous values*. https://www.dcceew.gov.au/sites/default/files/documents/factsheet-wetlands-indigenous-values.pdf

Thorp, J. R., & Lynch, R. (2000). *The determination of weeds of national significance*. National weeds strategy executive committee.

Thresher, R. E., Hayes, K., Bax, N. J., Teem, J., Benfey, T. J., & Gould, F. (2014). Genetic control of invasive fish: Technological options and its role in integrated pest management. *Biological Invasions*, *16*(6), 1201–1216. https://doi.org/10.1007/s10530-013-0477-0

Thresher, R. E., Jones, M., & Drake, D. A. R. (2019). Evaluating active genetic options for the control of sea lamprey (*Petromyzon marinus*) in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences*, *76*(7), 1186–1202. https://doi.org/10.1139/cjfas-2018-0153

Thresher, R. E., van de Kamp, J., Campbell, G., Grewe, P., Canning, M., Barney, M., Bax, N. J., Dunham, R., Su, B., & Fulton, W. (2014). Sex-ratio-biasing constructs for the control of invasive lower vertebrates. *Nature Biotechnology*, *32*(5), Article 5. https://doi.org/10.1038/nbt.2903

Thresher, R. E., Werner, M., Høeg, J. T., Svane, I., Glenner, H., Murphy, N. E., & Wittwer, C. (2000). Developing the options for managing marine pests: Specificity trials on the parasitic castrator, *Sacculina carcini*, against the European crab, *Carcinus maenas*, and related species. *Journal of Experimental Marine Biology and Ecology*, *254*(1), 37–51. https://doi.org/10.1016/S0022-0981(00)00260-4

Ticktin, T., Whitehead, A. N., & Fraiola, H. (2006). Traditional gathering of native hula plants in alien-invaded Hawaiian forests: Adaptive practices, impacts on alien invasive species and conservation implications. *Environmental Conservation*, *33*(3), 185–194. JSTOR. https://www.jstor.org/stable/44521908

Tildesley, M. J., Savill, N. J., Shaw, D. J., Deardon, R., Brooks, S. P., Woolhouse, M. E. J., Grenfell, B. T., & Keeling, M. J. (2006). Optimal reactive vaccination strategies for a foot-and-mouth outbreak in the UK. *Nature*, *440*(7080), 83–86. https://doi.org/10.1038/nature04324

Tildesley, M. J., Smith, G., & Keeling, M. J. (2012). Modeling the spread and control of foot-and-mouth disease in Pennsylvania following its discovery and options for control. *Preventive Veterinary Medicine*, *104*(3), 224–239. https://doi.org/10.1016/j.prevetmed.2011.11.007

Timmins, S. M., & Popay, A. I. (2002). The scientific approach to managing New Zealand's environmental weeds. *Defending the Green Oasis: New Zealand Biosecurity and Science: Proceedings of the New Zealand Plant Protection Society Symposium, Rotorua August*, 101–108.

Tobin, P. C., Kean, J. M., Suckling, D. M., McCullough, D. G., Herms, D. A., & Stringer, L. D. (2014). Determinants of successful arthropod eradication programs. *Biological Invasions*, *16*(2), 401–414. https://doi.org/10.1007/s10530-013-0529-5

Tolentino, H., Kamadjeu, R., Fontelo, P., Liu, F., Matters, M., Pollack, M., & Madoff, L. (2007). Scanning the Emerging Infectious Diseases Horizon—Visualizing ProMED Emails Using EpiSPIDER. *Advances in Disease Surveillance*, *2*, 169. https://lhncbc.nlm.nih.gov/publication/lhncbc-2007-055

Toral-Granda, M. V., Causton, C. E., Jäger, H., Trueman, M., Izurieta, J. C., Araujo, E., Cruz, M., Zander, K. K., Izurieta, A., & Garnett, S. T. (2017). Alien species pathways to the Galapagos Islands, Ecuador. *PLoS ONE*, *12*(9), e0184379. https://doi.org/10.1371/journal.pone.0184379

Towns, D. R., Daugherty, C. H., Broome, K., Timmins, S., & Clout, M. (2019). The thirty-year conservation revolution in New Zealand: An introduction. *Journal of the Royal Society of New Zealand*, *49*(3), 243–258. https://doi.org/10.1080/03036758.2019.1652192

Tracey, J., Lane, C., Fleming, P., Dickman, C., Quinn, J., Buckmaster, T., & McMahon, S. (2015). *2015 National Feral Cat Management Workshop Proceedings*. Invasive Animals Cooperative Research Centre. https://www.pestsmart.org.au/wp-content/uploads/2015/09/2015CatWorkshop_Proceedings_FINAL.pdf

Trouvé, R., & Robinson, A. P. (2021). Estimating Consignment-Level Infestation Rates from the Proportion of Consignment that Failed Border Inspection: Possibilities and Limitations in the Presence of Overdispersed Data. *Risk Analysis*, *41*(6), 992–1003. https://doi.org/10.1111/risa.13592

Trowbridge, C. C., Stanley, A., Kaye, T. N., Dunwiddie, P. W., & Williams, J. L. (2017). Long-term effects of prairie restoration on plant community structure and native population dynamics. *Restoration Ecology*, *25*(4), 559–568. https://doi.org/10.1111/rec.12468

Truelove, N. K., Patin, N. V., Min, M., Pitz, K. J., Preston, C. M., Yamahara, K. M., Zhang, Y., Raanan, B. Y., Kieft, B., Hobson, B., Thompson, L. R., Goodwin, K. D., & Chavez, F. P. (2022). Expanding the temporal and spatial scales of environmental DNA research with autonomous sampling. *Environmental DNA*, *4*(4), 972–984. https://doi.org/10.1002/edn3.299

Tucker, K. C., & Richardson, D. M. (1995). An expert system for screening potentially invasive alien plants in South African fynbos. *Journal of Environmental Management*, *44*(4), 309–338. https://doi.org/10.1016/S0301-4797(95)90347-X

Twigg, L. E., Martin, G. R., & Lowe, T. J. (2002). Evidence of pesticide resistance in medium-sized mammalian pests: A case study with 1080 poison and Australian rabbits. *Journal of Applied Ecology*, *39*(4), 549–560. https://doi.org/10.1046/j.1365-2664.2002.00738.x

UNEP. (2011). *The strategic plan for biodiversity 2011–2020 and the Aichi biodiversity targets*. UNEP/CBD/COP/DEC/X/2 29 October 2010, Nagoya, Japan. COP CBD Tenth Meeting. www.cbd.int/decisions/cop/?m=cop-10

Upadhyay, B., Burra, D. D., Nguyen, T. T., & Wyckhuys, K. A. G. (2020). Caught off guard: Folk knowledge proves deficient when addressing invasive pests in Asian cassava systems. *Environment, Development and Sustainability*, *22*(1), 425–445. https://doi.org/10.1007/s10668-018-0208-x

UPGE. (2020). *Préconisations pour une meilleure prise en compte du risque de dissémination des espèces végétales exotiques envahissantes (EVEE) terrestres dans les projets de travaux.* https://www.genieecologique.fr/reference-biblio/preconisations-pour-une-meilleure-prise-en-compte-du-risque-de-dissemination-des

Usher, M. B. (1988). Biological invasions of nature reserves: A search for generalisations. *Biological Conservation*, *44*(1–2), 119–135. https://doi.org/10.1016/0006-3207(88)90007-9

Usseglio, P., Selwyn, J. D., Downey-Wall, A. M., & Hogan, J. D. (2017). Effectiveness of removals of the invasive lionfish: How many dives are needed to deplete a reef? *PeerJ*, *5*, e3043. https://doi.org/10.7717/peerj.3043

van de Kant, K. D., van der Sande, L. J., Jöbsis, Q., van Schayck, O. C., & Dompeling, E. (2012). Clinical use of exhaled volatile organic compounds in pulmonary diseases: A systematic review. *Respiratory Research*, *13*(1), 117. https://doi.org/10.1186/1465-9921-13-117

van der Bles, A. M. (2019). Be certainly uncertain. *New Scientist*, *243*(3237), 21. https://doi.org/10.1016/S0262-4079(19)31224-2

Van der Colff, D., Kumschick, S., Foden, W., & Wilson, J. R. U. (2020). Comparing the IUCN's EICAT and Red List to improve assessments of the impact of biological invasions. *NeoBiota*, *62*, 509–523. https://doi.org/10.3897/neobiota.62.52623

Van Driesche, R. G., Carruthers, R. I., Center, T., Hoddle, M. S., Hough-Goldstein, J., Morin, L., Smith, L., Wagner, D. L., Blossey, B., Brancatini, V., Casagrande, R., Causton, C. E., Coetzee, J. A., Cuda, J., Ding, J., Fowler, S. V., Frank, J. H., Fuester, R., Goolsby, J., … van Klinken, R. D. (2010). Classical biological control for the protection of natural ecosystems. *Biological Control*, *54*(Supplement 1), S2–S33. https://doi.org/10.1016/j.biocontrol.2010.03.003

van Eeden, L. M., Newsome, T. M., Crowther, M. S., Dickman, C. R., & Bruskotter, J. (2020). Diverse public perceptions of species' status and management align with conflicting conservation frameworks. *Biological Conservation*, *242*, 108416. https://doi.org/10.1016/j.biocon.2020.108416

van Kleunen, M., Essl, F., Pergl, J., Brundu, G., Carboni, M., Dullinger, S., Early, R., González-Moreno, P., Groom, Q. J., Hulme, P. E., Kueffer, C., Kühn, I., Máguas, C., Maurel, N., Novoa, A., Parepa, M., Pyšek, P., Seebens, H., Tanner, R., … Dehnen-Schmutz, K. (2018). The changing role of ornamental horticulture in alien plant invasions. *Biological Reviews*, *93*(3), 1421–1437. https://doi.org/10.1111/brv.12402

van Klinken, R. D., Fiedler, K., Kingham, L., Collins, K., & Barbour, D. (2020). A risk framework for using systems approaches to manage horticultural biosecurity risks for market access. *Crop Protection*, *129*, 104994. https://doi.org/10.1016/j.cropro.2019.104994

van Klinken, R. D., Morin, L., Sheppard, A., & Raghu, S. (2016). Experts know more than just facts: Eliciting functional understanding to help prioritise weed biological control targets. *Biological Invasions*, *18*(10), 2853–2870. https://doi.org/10.1007/s10530-016-1175-5

van Rees, C. B., Hand, B. K., Carter, S. C., Bargeron, C., Cline, T. J., Daniel, W., Ferrante, J. A., Gaddis, K., Hunter, M. E., Jarnevich, C. S., McGeoch, M. A., Morisette, J. T., Neilson, M. E., Roy, H. E., Rozance, M. A., Sepulveda, A., Wallace, R. D., Whited, D., Wilcox, T., … Luikart, G. (2022). A framework to integrate innovations in invasion science for proactive management. *Biological Reviews*, *97*(4), 1712–1735. https://doi.org/10.1111/brv.12859

van Wilgen, B. W. (2012). Evidence, perceptions, and trade-offs associated with invasive alien plant control in the Table Mountain National Park, South Africa. *Ecology and Society*, *17*(2), 23. https://doi.org/10.5751/ES-04590-170223 van Wilgen, B. W., De Wit, M. P., Anderson, H. J., Le Maitre, D. C., Kotze, I. M., Ndala, S., Brown, B., & Rapholo, M. B. (2004). Costs and benefits of biological control of invasive alien plants: Case studies from South Africa. *South African Journal of Science*, *100*(1–2), 113–122. https://www.dws.gov.za/wfw/docs/VanWilgenetal.,2004.pdf

van Wilgen, B. W., Forsyth, G. G., Le Maitre, D. C., Wannenburgh, A., Kotzé, J. D. F., van den Berg, E., & Henderson, L. (2012). An assessment of the effectiveness of a large, national-scale invasive alien plant control strategy in South Africa. *Biological Conservation*, *148*(1), 28–38. https://doi.org/10.1016/j.biocon.2011.12.035

van Wilgen, B. W., & Richardson, D. M. (2014). Challenges and trade-offs in the management of invasive alien trees. *Biological Invasions*, *16*(3), 721–734. https://doi.org/10.1007/s10530-013-0615-8

van Wilgen, B. W., & Wannenburgh, A. (2016). Co-facilitating invasive species control, water conservation and poverty relief: Achievements and challenges in South Africa's Working for Water programme. *Current Opinion in Environmental Sustainability*, *19*, 7–17. https://doi.org/10.1016/j.cosust.2015.08.012

van Wilgen, B. W., Wilson, J. R., Wannenburgh, A., & Foxcroft, L. C. (2020). The Extent and Effectiveness of Alien Plant Control Projects in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 597–628). Springer International Publishing. https://doi.org/10.1007/978-3-030-32394-3_21

van Zalinger, R. (2006). *An assessment of exotic species in the Tonle Sap Biosphere Reserve and associated threats to biodiversity. A resource document for the management of invasive alien species.* (p. 96). UNDP/GEF-funded Tonle Sap Conservation Project. https://library.wcs.org/doi/ctl/view/mid/33065/pubid/DMX1252900000.aspx

Vanderhoeven, S., Adriaens, T., D'hondt, B., Van Gossum, H., Vandegehuchte, M., Verreycken, H., Cigar, J., & Branquart, E. (2015). A science-based approach to tackle invasive alien species in Belgium – the role of the ISEIA protocol and the Harmonia information system as decision support tools. *Management of Biological Invasions*, *6*(2), 197–208. https://doi.org/10.3391/mbi.2015.6.2.10

Vanderhoeven, S., Branquart, E., Casaer, J., D'hondt, B., Hulme, P. E., Shwartz, A., Strubbe, D., Turbé, A., Verreycken, H., & Adriaens, T. (2017). Beyond protocols: Improving the reliability of expert-based risk analysis underpinning invasive species policies. *Biological Invasions*, *19*(9), 2507–2517. https://doi.org/10.1007/s10530-017-1434-0

Vannatta, A. R., Hauer, R. H., & Schuettpelz, N. M. (2012). Economic Analysis of Emerald Ash Borer (Coleoptera: Buprestidae) Management Options. *Journal of Economic Entomology*, *105*(1), 196–206. https://doi.org/10.1603/EC11130

Veitch, C. R., & Clout, M. N. (Eds.). (2002). *Turning the Tide: The Eradication of Invasive Species : Proceedings of the International Conference on Eradication of Island Invasives*. IUCN SSC Invasive Species Specialist Group.

Veitch, C. R., Clout, M. N., & Towns, D. R. (2011). *Island Invasives: Eradication and Management Proceedings of the International Conference on Island Invasives*. https://policycommons.net/artifacts/1375054/island-invasives/1989310/

Venette, R. C. (2015). The challenge of modelling and mapping the future distribution and impact of invasive alien species. In *Pest risk modelling and mapping for invasive alien species* (USDA, pp. 1–17). CABI. https://doi.org/10.1079/9781780643946.0001

Venette, R. C., Kriticos, D. J., Magarey, R. D., Koch, F. H., Baker, R. H. A., Worner, S. P., Gómez Raboteaux, N. N., McKenney, D. W., Dobesberger, E. J., Yemshanov, D., De Barro, P. J., Hutchison, W. D., Fowler, G., Kalaris, T. M., & Pedlar, J. (2010). Pest Risk Maps for Invasive Alien Species: A Roadmap for Improvement. *BioScience*, *60*(5), 349–362. https://doi.org/10.1525/bio.2010.60.5.5

Vimercati, G., Davies, S. J., Hui, C., & Measey, J. (2017). Does restricted access limit management of invasive urban frogs? *Biological Invasions*, *19*(12), 3659–3674. https://doi.org/10.1007/s10530-017-1599-6

Vimercati, G., Kumschick, S., Probert, A. F., Volery, L., & Bacher, S. (2020). The importance of assessing positive and beneficial impacts of alien species. *NeoBiota*, *62*, 525–545. https://doi.org/10.3897/neobiota.62.52793

Vishnu Chandran, M., & Gopakumar, S. (Guide). (2018). *Impact of invasive alien plants on understorey vegetation in Wayanad wildlife sanctuary* [Department of Natural Resource Management College of Forestry]. http://hdl.handle.net/123456789/8047

Vissoh, P. V., Gbèhounou, G., Ahanchédé, A., Kuyper, T. W., & Röling, N. G. (2004). Weeds as agricultural constraint to farmers in Benin: Results of a diagnostic study. *NJAS – Wageningen Journal of Life Sciences*, *52*(3–4), 305–329. https://doi.org/10.1016/S1573-5214(04)80019-8

Vítková, M., Müllerová, J., Sádlo, J., Pergl, J., & Pyšek, P. (2017). Black locust (*Robinia pseudoacacia*) beloved and despised: A story of an invasive tree in Central Europe. *Forest Ecology and Management*, *384*, 287–302. https://doi.org/10.1016/j.foreco.2016.10.057

Vogel, E., Santos, D., Mingels, L., Verdonckt, T.-W., & Broeck, J. V. (2019). RNA Interference in Insects: Protecting Beneficials and Controlling Pests. *Frontiers in Physiology*, *9*. https://www.frontiersin.org/articles/10.3389/fphys.2018.01912

Volery, L., Blackburn, T. M., Bertolino, S., Evans, T., Genovesi, P., Kumschick, S., Roy, H. E., Smith, K. G., & Bacher, S. (2020). Improving the Environmental Impact Classification for Alien Taxa (EICAT): A summary of revisions to the framework and guidelines. *NeoBiota*, *62*, 547–567. https://doi.org/10.3897/neobiota.62.52723

Vranjic, J. A., Morin, L., Reid, A. M., & Groves, R. H. (2012). Integrating revegetation with management methods to rehabilitate coastal vegetation invaded by Bitou bush (*Chrysanthemoides monilifera* ssp. *Rotundata*) in Australia. *Austral Ecology*, *37*(1), 78–89. https://doi.org/10.1111/j.1442-9993.2011.02242.x

Wadsworth, R. A., Collingham, Y. C., Willis, S. G., Huntley, B., & Hulme, P. E. (2000). Simulating the spread and management of alien riparian weeds: Are they out of control? *Journal of Applied Ecology*, *37*(s1), 28–38. https://doi.org/10.1046/j.1365-2664.2000.00551.x

Wäldchen, J., Rzanny, M., Seeland, M., & Mäder, P. (2018). Automated plant species identification—Trends and future directions. *PLOS Computational Biology*, *14*(4), e1005993. https://doi.org/10.1371/journal.pcbi.1005993

Wallace, R. D., Bargeron, C. T., & Reaser, J. K. (2020). Enabling decisions that make a difference: Guidance for improving access to and analysis of invasive species information. *Biological Invasions*, *22*(1), 37–45. https://doi.org/10.1007/s10530-019-02142-2

Walsh, S. J. (2018). Multi-scale Remote Sensing of Introduced and Invasive Species: An Overview of Approaches and Perspectives. In M. de L. Torres & C. F. Mena (Eds.), *Understanding Invasive Species in the Galapagos Islands: From the Molecular to the Landscape* (pp. 143–154). Springer International Publishing. https://doi.org/10.1007/978-3-319-67177-2_8

Walter, K. J., & Armstrong, K. V. (2014). Benefits, threats and potential of *Prosopis* in South India. *Forests, Trees and Livelihoods*, *23*(4), 232–247. https://doi.org/10.1080/14728028.2014.919880

Walther, G.-R., Roques, A., Hulme, P. E., Sykes, M. T., Pyšek, P., Kühn, I., Zobel, M., Bacher, S., Botta-Dukát, Z., Bugmann, H., Czúcz, B., Dauber, J., Hickler, T., Jarošík, V., Kenis, M., Klotz, S., Minchin, D., Moora, M., Nentwig, W., … Settele, J. (2009). Alien species in a warmer world: Risks and opportunities. *Trends in Ecology & Evolution*, *24*(12), 686–693. https://doi.org/10.1016/j.tree.2009.06.008

Wan, F., Guo, J. Y., & Zhang, F. (2009). *Research on biological invasions in China.* Science Press. https://www.cabdirect.org/cabdirect/abstract/20103059337

Wan, F., Jiang, M., & Zhan, A. (Eds.). (2017). *Biological invasions and its management in China*. Springer.

Wang, X., Walton, J. R., Parshad, R. D., Storey, K., & Boggess, M. (2014). Analysis of the *Trojan Y-Chromosome* eradication strategy for an invasive species. *Journal of Mathematical Biology*, *68*(7), 1731–1756. https://doi.org/10.1007/s00285-013-0687-1

Wanzala, W., Takken, W., Mukabana, W. R., Pala, A. O., & Hassanali, A. (2012). Ethnoknowledge of Bukusu community on livestock tick prevention and control in Bungoma district, western Kenya. *Journal of Ethnopharmacology*, *140*(2), 298–324. https://doi.org/10.1016/j.jep.2012.01.021

Ward, A. I., Richardson, S., Macarthur, R., & Mill, A. C. (2020). Using and communicating uncertainty for the effective control of invasive non-native species. *Mammal Review*, *50*(2), 211–220. https://doi.org/10.1111/mam.12188

Ward, M. P., Tian, K., & Nowotny, N. (2021). African Swine Fever, the forgotten pandemic. *Transboundary and Emerging Diseases*, *68*(5), 2637–2639. https://doi.org/10.1111/tbed.14245

Waterhouse, D. F., & Sands, D. P. A. (2001). *Classical biological control of arthropods in Australia. ACIAR monograph No. 77*. CSIRO Entomology, Australian Centre for International Agricultural Research.

Webber, B. L., & Scott, J. K. (2012). Rapid global change: Implications for defining natives and aliens. *Global Ecology and Biogeography*, *21*(3), 305–311. https://doi.org/10.1111/j.1466-8238.2011.00684.x

Weiberg, A., Wang, M., Lin, F.-M., Zhao, H., Zhang, Z., Kaloshian, I., Huang, H.-D., & Jin, H. (2013). Fungal Small RNAs Suppress Plant Immunity by Hijacking Host RNA Interference Pathways. *Science*, *342*(6154), 118–123. https://doi.org/10.1126/science.1239705

Weidlich, E. W. A., Flórido, F. G., Sorrini, T. B., & Brancalion, P. H. S. (2020). Controlling invasive plant species in ecological restoration: A global review. *Journal of Applied Ecology*, *57*(9), 1806–1817. https://doi.org/10.1111/1365-2664.13656

Weir, J., & Duff, N. (2015). Weeds and Native Title in the Kimberley. In E. Ens, J. Fisher, & O. Costello (Eds.), *Indigenous people and invasive species: Perceptions, management, challenges and uses. IUCN Commission on Ecosystem Management Community Report.* (pp. 21–22).

Weiss, D. J., Nelson, A., Gibson, H. S., Temperley, W., Peedell, S., Lieber, A., Hancher, M., Poyart, E., Belchior, S., Fullman, N., Mappin, B., Dalrymple, U., Rozier, J., Lucas, T. C. D., Howes, R. E., Tusting, L. S., Kang, S. Y., Cameron, E., Bisanzio, D., … Gething, P. W. (2018). A global map of travel time to cities to assess inequalities in accessibility in 2015. *Nature*, *553*(7688), 333–336. https://doi.org/10.1038/nature25181

Wells, C. R., Sah, P., Moghadas, S. M., Pandey, A., Shoukat, A., Wang, Y., Wang, Z., Meyers, L. A., Singer, B. H., & Galvani, A. P. (2020). Impact of international travel and border control measures on the global spread of the novel 2019 coronavirus outbreak. *Proceedings of the National Academy of Sciences*, *117*(13), 7504–7509. https://doi.org/10.1073/pnas.2002616117

Wells, F. E. (2019). Environmental Emergency: Why Did the False Mussel *Mytilopsis sallei* Not Invade Darwin Harbour, Australia? *Malacologia*, *62*(2), 247–256. https://doi.org/10.4002/040.062.0205

Welvaert, M., & Caley, P. (2016). Citizen surveillance for environmental monitoring: Combining the efforts of citizen science and crowdsourcing in a quantitative data framework. *SpringerPlus*, *5*(1), 1890. https://doi.org/10.1186/s40064-016-3583-5

Weng, X., Kang, Y., Guo, Q., Peng, B., & Jiang, H. (2019). Recent advances in thread-based microfluidics for diagnostic applications. *Biosensors and Bioelectronics*, *132*, 171–185. https://doi.org/10.1016/j.bios.2019.03.009

Werschkun, B., Banerji, S., Basurko, O. C., David, M., Fuhr, F., Gollasch, S., Grummt, T., Haarich, M., Jha, A. N., Kacan, S., Kehrer, A., Linders, J., Mesbahi, E., Pughiuc, D., Richardson, S. D., Schwarz-Schulz, B., Shah, A., Theobald, N., von Gunten, U., … Höfer, T. (2014). Emerging risks from ballast water treatment: The run-up to the International Ballast Water Management Convention. *Chemosphere*, *112*, 256–266. https://doi.org/10.1016/j.chemosphere.2014.03.135

Westwood, J. H., Charudattan, R., Duke, S. O., Fennimore, S. A., Marrone, P., Slaughter, D. C., Swanton, C., & Zollinger, R. (2018). Weed Management in 2050: Perspectives on the Future of Weed Science. *Weed Science*, *66*(3), 275–285. https://doi.org/10.1017/wsc.2017.78

Weyl, O. L. F., Ellender, B. R., Wasserman, R. J., & Woodford, D. J. (2015). Unintended consequences of using alien fish for human benefit in protected areas. *Koedoe*, *57*(1), a1264. https://doi.org/10.4102/koedoe.v57i1.1264

Weyl, O. L. F., Ellender, B. R., Woodford, D. J., & Jordaan, M. S. (2013). Fish distributions in the Rondegat River, Cape Floristic Region, South Africa, and the immediate impact of rotenone treatment in an invaded reach. *African Journal of Aquatic Science*, *38*(2), 201–209. https://doi.org/10.2989/16085914.2012.753401

Weyl, P. S. R., de Moor, F. C., Hill, M. P., & Weyl, O. L. F. (2010). The effect of largemouth bass *Micropterus salmoides* on aquatic macro-invertebrate communities in the Wit River, Eastern Cape, South Africa. *African Journal of Aquatic Science*, *35*(3), 273–281. https://doi.org/10.2989/16085914.2010.540776

WHO. (2010). *International Code of Conduct on the Distribution and Use of Pesticides Guidelines for the Registration of Pesticides*. World Health Organization. https://apps.who.int/iris/bitstream/handle/10665/70293/WHO_HTM_NTD_WHOPES_2010.7_eng.pdf

WHO. (2013). *Health and environment: Communicating the risks*. World Health Organization.https://apps.who.int/iris/bitstream/handle/10665/108629/9789289000512-eng.pdf?sequence=1&isAllowed=y

Whyard, S., Erdelyan, C. N., Partridge, A. L., Singh, A. D., Beebe, N. W., & Capina, R. (2015). Silencing the buzz: A new approach to population suppression of mosquitoes by feeding larvae double-stranded RNAs. *Parasites & Vectors*, *8*(1), 96. https://doi.org/10.1186/s13071-015-0716-6

Whyte, C. (2006). Science and biosecurity–monitoring the effectiveness of biosecurity interventions at New Zealand's borders. *Royal Society of New Zealand, Miscellaneous Series*, *67*, 27–36.

Willan, R. C., Russell, B. C., Murfet, N. B., Moore, K. L., McEnnulty, F. R., Horner, S. K., Hewitt, C. L., Dally, G. M., Campbell, M. L., & Bourke, S. T. (2000). Outbreak of *Mytilopsis sallei* (Récluz, 1849) (Bivalvia: Dreissenidae) in Australia. *Molluscan Research*, *20*(2), 25–30. https://doi.org/10.1080/13235818.2000.10673730

Willis, A. J., & Memmott, J. (2005). The potential for indirect effects between a weed, one of its biocontrol agents and native herbivores: A food web approach. *Biological Control*, *35*(3), 299–306. https://doi.org/10.1016/j.biocontrol.2005.07.013

Wilson, A. D. (2017). Electronic-nose devices—Potential for noninvasive early disease-detection applications. *Annals of Clinical Case Reports*, *2*, Article 1401. https://www.fs.usda.gov/treesearch/pubs/54652

Wilson, J. R. U., Bacher, S., Daehler, C. C., Groom, Q. J., Kumschick, S., Lockwood, J. L., Robinson, T. B., Zengeya, T. A., & Richardson, D. M. (2020). Frameworks used in invasion science: Progress and prospects. *NeoBiota*, *62*, 1–30. https://doi.org/10.3897/neobiota.62.58738

Wilson, J. R. U., Ivey, P., Manyama, P., & Nänni, I. (2013). A new national unit for invasive species detection, assessment and eradication planning. *South African Journal of Science*, *109*(5/6), 1–13. https://doi.org/10.1590/sajs.2013/20120111

Wilson, J. R. U., Panetta, F. D., & Lindgren, C. (2016). *Detecting and Responding to Alien Plant Incursions*. Cambridge University Press. https://scholar.google.com.au/scholar?hl=en&as_sdt=0%2C5&q=Detecting+and+responding+to+alien+plant+incursions&btnG=

Wilson, K. A., Underwood, E. C., Morrison, S. A., Klausmeyer, K. R., Murdoch, W. W., Reyers, B., Wardell-Johnson, G., Marquet, P. A., Rundel, P. W., McBride, M. F., Pressey, R. L., Bode, M., Hoekstra, J. M., Andelman, S., Looker, M., Rondinini, C., Kareiva, P., Shaw, M. R., & Possingham, H. P. (2007). Conserving biodiversity efficiently: What to do, where, and when. *PLoS Biology*, *5*(9), e223. https://doi.org/10.1371/journal.pbio.0050223

Wilson, M. E., & Coulson, G. (2016). Comparative efficacy of levonorgestrel and deslorelin contraceptive implants in free-ranging eastern grey kangaroos (*Macropus giganteus*). *Wildlife Research*, *43*(3), 212–219. https://doi.org/10.1071/WR15176

Winston, R. L., Schwarzlander, M., Hinz, H. L., Day, M. D., Cock, M. J. W., & Julien, M. H. (Eds.). (2014). *Biological control of weeds: A world catalogue of agents and their target weeds, 5th edn.* USDA Forest Service, Forest Health Technology Enterprise Team, Morgantown. https://www.ibiocontrol.org/catalog/about.pdf

Witt, A. B. R., Kiambi, S., Beale, T., & Van Wilgen, B. W. (2017). A preliminary assessment of the extent and potential impacts of alien plant invasions in the Serengeti-Mara ecosystem, East Africa. *Koedoe*, *59*(1), a1426. https://doi.org/10.4102/koedoe.v59i1.1426

Wittenberg, R., & Cock, M. J. W. (2003). *Invasive Alien Species: A Toolkit of Best Prevention and Management Practices*. CAB International. https://www.cabidigitallibrary.org/doi/10.1079/9780851995694.0000

Woolnough, A. P., Hampton, J. O., Campbell, S., Lethbridge, M. R., Boardman, W. S. J., Sharp, T., & Rose, K. (2012). Field Immobilization Of Feral 'Judas' Donkeys (*Equus asinus*) By Remote Injection Of Medetomidine And Ketamine And Antagonism With Atipamezole. *Journal of Wildlife Diseases*, *48*(2), 435–443. https://doi.org/10.7589/0090-3558-48.2.435

World Organisation for Animal Health. (2019). Chapter 1.4. Animal Health Surveillance. In *Terrestrial Animal Health Code*. World Organisation for Animal Health. https://www.oie.int/index.php?id=169&L=0&htmfile=chapitre_surveillance_general.htm

World Organisation for Animal Health. (2020, June 18). Overview of the activities of the OIE Specialist Commissions. *OIE – World Organisation for Animal Health*. https://www.oie.int/en/overview-of-the-activities-of-the-oie-specialist-commissions/

Xiao, D. R., Zhang, L. Q., Zhu, Z. C., & Tian, K. (2011). *Response of seed production and viability of* Spartina alternifolia *to cutting at Shanghai Chongming DongTan, China. 20*, 1681–1686. http://www.alljournals.cn/view_abstract.aspx?pcid=3FF3ABA7486768130C3FF830376F43B398E0C97F0FF2DD53&cid=BEC1F4700DD66C14D137590A0896A0AD&jid=1CB67BEB88F124F56F6BC6645431F032&aid=42BF0787407D9153CB0AD4178164EB22&yid=9377ED8094509821

Yainna, S., Nègre, N., Silvie, P. J., Brévault, T., Tay, W. T., Gordon, K., dAlençon, E., Walsh, T., & Nam, K. (2021). Geographic Monitoring of Insecticide Resistance Mutations in Native and Invasive Populations of the Fall Armyworm. *Insects*, *12*(5), Article 5. https://doi.org/10.3390/insects12050468

Yan, Z., Sun, J., Don, O., & Zhang, Z. (2005). The red turpentine beetle, *Dendroctonus valens* LeConte (Scolytidae): An exotic invasive pest of pine in China. *Biodiversity & Conservation*, *14*(7), 1735–1760. https://doi.org/10.1007/s10531-004-0697-9

Ye, X., Li, L., Li, J., Wu, X., Fang, X., & Kong, J. (2019). Microfluidic-CFPA Chip for the Point-of-Care Detection of African Swine Fever Virus with a Median Time to Threshold in about 10 min. *ACS Sensors*, *4*(11), 3066–3071. https://doi.org/10.1021/acssensors.9b01731

Yemshanov, D., Koch, F. H., & Ducey, M. (2015). Making Invasion Models Useful for Decision Makers: Incorporating Uncertainty, Knowledge Gaps and Decision-making Preferences. In R. C. Venette (Ed.), *Pest risk modelling and mapping for Invasive Alien Species* (pp. 206–222). CABI International and USDA.

Yesson, C., Brewer, P. W., Sutton, T., Caithness, N., Pahwa, J. S., Burgess, M., Gray, W. A., White, R. J., Jones, A. C., Bisby, F. A., & Culham, A. (2007). How Global Is the Global Biodiversity Information Facility? *PLoS ONE*, *2*(11), e1124. https://doi.org/10.1371/journal.pone.0001124

Yick, J. L., Wisniewski, C., Diggle, J., & Patil, J. G. (2021). Eradication of the Invasive Common Carp, *Cyprinus carpio* from a Large Lake: Lessons and Insights from the Tasmanian Experience. *Fishes*, *6*(1), 6. https://doi.org/10.3390/fishes6010006

Yu, V. L., & Madoff, L. C. (2004). ProMED-mail: An Early Warning System for Emerging Diseases. *Clinical Infectious Diseases*, *39*(2), 227–232. https://doi.org/10.1086/422003

Zabala, J., Zuberogoitia, I., & González-Oreja, J. A. (2010). Estimating costs and outcomes of invasive American mink (*Neovison vison*) management in continental areas: A framework for evidence based control and eradication. *Biological Invasions*, *12*(9), 2999–3012. https://doi.org/10.1007/s10530-010-9690-2

Zachrisson, B., & Barba, A. (2020). Biological Control in Panama. In *Biological Control in Latin America and the Caribbean: Its Rich History and Bright Future* (pp. 345–353). CABI International. https://doi.org/10.1079/9781789242430.0345

Zahid, I., Grgurinovic, C., Zaman, T., De Keyzer, R., & Cayzer, L. (2012). Assessment of technologies and dogs for detecting insect pests in timber and forest products. *Scandinavian Journal of Forest Research*, *27*(5), 492–502. https://doi.org/10.1080/02827581.2012.657801

Zahra, S., Hofstetter, R. W., Waring, K. M., & Gehring, C. (2020). Review: The invasion of *Acacia nilotica* in Baluran National Park, Indonesia, and potential future control strategies. *Biodiversitas Journal of Biological Diversity*, *21*(1), Article 1. https://doi.org/10.13057/biodiv/d210115

Zaiko, A., Samuiloviene, A., Ardura, A., & Garcia-Vazquez, E. (2015). Metabarcoding approach for nonindigenous species surveillance in marine coastal waters. *Marine Pollution Bulletin*, *100*(1), 53–59. https://doi.org/10.1016/j.marpolbul.2015.09.030

Zalba, S., & Ziller, S. R. (2007). Adaptive management of alien invasive species: Putting the theory into practice. *Natureza & Conservação*, *5*(2), 86–92. https://www.cabdirect.org/cabdirect/abstract/20073250023

Zarco-Tejada, P. J., Camino, C., Beck, P. S. A., Calderon, R., Hornero, A., Hernández-Clemente, R., Kattenborn, T., Montes-Borrego, M., Susca, L., Morelli, M., Gonzalez-Dugo, V., North, P. R. J., Landa, B. B., Boscia, D., Saponari, M., & Navas-Cortes, J. A. (2018). Previsual symptoms of *Xylella fastidiosa* infection revealed in spectral plant-trait alterations. *Nature Plants*, *4*(7), 432–439. https://doi.org/10.1038/s41477-018-0189-7

Zavaleta, E. S. (2000). The Economic Value of Controlling an Invasive Shrub. *AMBIO: A Journal of the Human Environment*, *29*(8), 462–467. https://doi.org/10.1579/0044-7447-29.8.462

Zavaleta, E. S., Hobbs, R. J., & Mooney, H. A. (2001). Viewing invasive species removal in a whole-ecosystem context. *Trends in Ecology & Evolution*, *16*(8), 454–459. https://doi.org/10.1016/S0169-5347(01)02194-2

Zenni, R. D., Ziller, S. R., Pauchard, A., Rodriguez-Cabal, M., & Nuñez, M. A. (2017). Invasion Science in the Developing World: A Response to Ricciardi *et al*. *Trends in Ecology & Evolution*, *32*(11), 807–808. https://doi.org/10.1016/j.tree.2017.08.006

Zhang, W., & Swinton, S. M. (2009). Incorporating natural enemies in an economic threshold for dynamically optimal pest management. *Ecological Modelling*, *220*(9), 1315–1324. https://doi.org/10.1016/j.ecolmodel.2009.01.027

Zhang, Z., Yokota, M., & Strüssmann, C. A. (2019). A periodic matrix population model to predict growth potential of the invasive Chinese mitten crab *Eriocheir sinensis* (H. Milne Edwards, 1853) (Decapoda: Brachyura: Varunidae). *Journal of Crustacean Biology*, *39*(1), 28–35. https://doi.org/10.1093/jcbiol/ruy090

Zhu, H., Fohlerová, Z., Pekárek, J., Basova, E., & Neužil, P. (2020). Recent advances in lab-on-a-chip technologies for viral diagnosis. *Biosensors and Bioelectronics*, *153*, 112041. https://doi.org/10.1016/j.bios.2020.112041

Ziller, S. R., de Sá Dechoum, M., & Dudeque Zenni, R. (2019). Predicting invasion risk of 16 species of eucalypts using a risk assessment protocol developed for Brazil. *Austral Ecology*, *44*(1), 28–35. https://doi.org/10.1111/aec.12649

Ziller, S. R., Dechoum, M. de S., Silveira, R. A. D., da Rosa, H. M., Motta, M. S., da Silva, L. F., Oliveira, B. C. M., & Zenni, R. D. (2020). A priority-setting scheme for the management of invasive non-native species in protected areas. *NeoBiota*, *62*, 591–606. https://doi.org/10.3897/neobiota.62.52633

Zimmerman, C., Jordan, M., Sargis, G., Smith, H., & Schwager, K. (2011). *An Invasive Plant Management Decision Analysis Tool* (Version 1.1). The Nature Conservancy. https://cipwg.uconn.edu/wp-content/uploads/sites/244/2013/12/IPMDAT.pdf

Zivin, J., Hueth, B. M., & Zilberman, D. (2000). Managing a Multiple-Use Resource: The Case of Feral Pig Management in California Rangeland. *Journal of Environmental Economics and Management*, *39*(2), 189–204. https://doi.org/10.1006/jeem.1999.1101

Chapter 6

GOVERNANCE AND POLICY OPTIONS FOR THE MANAGEMENT OF BIOLOGICAL INVASIONS[1]

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Chapter 6

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Chapter **6**

GOVERNANCE AND POLICY OPTIONS FOR THE MANAGEMENT OF BIOLOGICAL INVASIONS

EXECUTIVE SUMMARY

1 Despite some successes over the past decade, there has been limited progress towards meeting international goals and targets for biological invasions, such as Aichi Target 9 of the Strategic Plan for Biodiversity 2011-2020 and Target 15.8 of the 2030 Agenda for Sustainable Development *(well established)* {6.1.2} (Table 6.4). Many countries have little or no funding for activities to prevent or control biological invasions *(well established)* {6.1.3}, and most national invasive alien species targets lack sufficient ambition to substantively contribute to the achievement of Aichi Biodiversity Target 9 *(well established)* {6.1.2, 6.3.3}. Legislative and other policy instruments addressing biological invasions and their implementation vary greatly across countries *(well established)* {6.1.2} and sectors within countries *(well established)* {6.3.1.1, 6.3.3.1}. Data for assessing the effectiveness of pathway- and species-management are mostly unavailable *(well established)* {6.6.1.2}, highlighting the need for the generation of new and up-to-date information on the appropriate level of implementation and on the successes and failures of management interventions {6.6.1, 6.7.2.6}. Furthermore, there is limited dedicated interdisciplinary research on the governance for biological invasions in a broader environmental governance context *(well established)* {6.2.2, 6.3.3.3, 6.6.1.4}.

2 Effective governance for biological invasions can address policy gaps and limits, and improve policy coherence and its implementation *(well established)* {6.2.3, 6.7.2.3}. This could be achieved with a context-specific integrated governance approach for biological invasions, that focuses on coordinated and sequential implementation of strategic actions *(established but incomplete)* {6.2.4, 6.7.2}. Integrated governance can be achieved through robust institutions that are responsive to changing contexts, and strategies to ensure effective implementation of strategic actions *(well established)* {6.7.3} **(Figure 6.21)**. A conducive environment for integrated governance is equitable and respects different value systems and perspectives *(well established)* {6.7.3} **(Figure 6.21)**. This in turn promotes inclusive decision-making, shared efforts and commitments, the understanding of specific roles of all actors, catalyses the sharing of knowledge, data and resources, and promotes the development and implementation of multidisciplinary solutions *(well established)* {6.7.2} **(Figure 6.21)**. Integrated governance includes explicitly considering negotiation and trade-offs as an integral part of the process *(established but incomplete)* {6.2.2} **(Figure 6.21)**. By focusing on the relationships between the scales, levels of governance, sectors and stakeholders involved, the integrated governance approach identifies and addresses feedbacks, efficiencies and trade-offs in the management of biological invasions *(well established)* {6.2.4, 6.2.3.3, 6.7.2, 6.7.3} **(Table 6.7)**.

3 Multilateral coordination and cooperation are key for bringing about progress towards achieving invasive alien species goals and targets, and coherent, mutually supportive actions *(well established)* {6.2.3.3, 6.3.2.3, 6.4.4, 6.7.2.2}. Widespread and purposeful cooperation could improve the effectiveness of policy instruments to prevent and control invasive alien species *(well established)* {6.3.2.3, 6.7.2.2}. The need for cooperation emerges from the diversity of stakeholders and perspectives involved {6.4.1}; problems created by uncoordinated responses and little-considered trade-offs across sectors {6.3.1.1, 6.7.2.2}; the multiple geopolitical scales at which policy and management are needed {6.3.2}; widespread economic constraints in some regions {6.1.3}; and the interdependence between invasive alien species and other drivers of change in nature {6.3.1.3, 6.7.2.2} *(established but incomplete)*. Strategies for achieving such cooperation and for managing the collective action costs of widespread collaboration can include: enhancing coordination and collaboration across international and regional mechanisms {6.2.3.4, 6.7.4}; long-term resourcing, including developing the capacity needed, and commitment from governments and institutions at the highest levels {6.2.3.2, 6.5.1, 6.7.2.3} **(Table 6.1)**. Other options to accelerate progress include research on the relationships between actors and institutions {6.6.1}; engagement of the general public including awareness campaigns {6.2.3.3, 6.4.2.2, 6.6.2.1}; inclusion of Indigenous Peoples and local communities and recognition of their rights {6.4.3.1, 6.4.3.2, 6.6.1.5}, and information platforms to support decision-making that are developed and sustained for the long-term *(established but incomplete)* {6.2.3.1(3), 6.6.2}.

4 One of the most effective ways to manage biological invasions is to develop policy instruments that seek synergies between human health, agriculture, forestry, fisheries and environment sectors at national and international levels *(established but incomplete)* {6.1.2, 6.3.1.1}. Many national laws and regulations, as well as multilateral agreements, aimed at preventing the introduction of invasive alien species have been adopted *(well established)* {6.1.2}. They have jointly contributed to reducing the risk of invasive alien species' impacts on nature's contributions to people and good quality of life *(well established)* {6.1.2}. However, there are still gaps, limitations, and inconsistencies in the scope, taxonomic coverage, procedures and standards of current policy instruments both within countries and across regions *(well established)* {6.2.2}. Close collaboration between the different national agencies overseeing trade policy, agriculture and forestry, the environment and health can deliver a coordinated approach to biological invasions *(well established)* {6.3.1.1}. Existing approaches for achieving the necessary coordination (such as EcoHealth, One Health and One Biosecurity approaches) provide frameworks for cross-disciplinary thinking in support of the development and implementation of policies and policy instruments *(established but incomplete)* {6.3.1.1, 6.7.2.2}. While economic incentives can potentially enhance compliance with biosecurity protocols *(established but incomplete)* {6.5.2.1}; financial deterrents in the form of tariffs and penalty systems can also be used to prevent mismanagement of species introductions and the revenues they generate as means to fund incentives and/or government control programmes *(established but incomplete)* {6.5.2.2, 6.5.2.3, 6.5.3, 6.5.4, 6.5.6}. Governance for biological invasions would also benefit from the expansion of dedicated inter- and transdisciplinary research *(well established)* {6.2.4, 6.6.1.4}. Research on the impacts of invasive alien species across the health, agriculture, forestry, fisheries and environment sectors could support the development of coherent policy instruments *(established but incomplete)* {6.2.4} **(Table 6.2)**.

5 Implementation-focused national strategies and action planning for biological invasions, aligned with international regulatory frameworks, could stimulate action and help improve the effectiveness and efficiency of management efforts *(well established)* {6.2.3.2, 6.3.3.1, 6.7.2.3}. Implementation-focused strategies and action plans can provide enabling conditions for the successful governance of biological invasions, including coordination and collaboration across international and regional mechanisms *(established but incomplete)* {6.2.4, 6.7.3}, legal, regulatory and institutional frameworks {6.3.3.1}, market-based instruments that provide economic incentives and deterrents {6.5.2} and multisector inclusion {6.2.3.3, 6.3.1.1}. These national strategies could prioritize the measurement and monitoring of the resource inputs, processes, outputs and outcomes needed to improve implementation and accelerate progress towards meeting invasive alien species goals and targets at multiple levels of governance *(established but incomplete)* {6.2.3, 6.3.3.3, 6.6.2, 6.6.3, 6.7.2.6}. National strategies can define the governance models, policy instruments and support tools needed to ensure shared efforts and commitments, and understanding of the specific roles of all actors *(well established)* {6.2.3.2, 6.7.2.5} and include plans for the effective engagement across private and government stakeholders and Indigenous Peoples and local communities *(established but incomplete)* {6.4.3, 6.6.1.5, 6.7.2.4, 6.7.2.5}. They can also include market-based instruments to fund and promote activities to prevent and manage biological invasions in national budgets *(established but incomplete)* {6.5.2, 6.5.3, 6.5.4, 6.5.6} and ensure the efficient use of resources for biological invasions *(well established)* {6.2.3.2, 6.2.3.4, 6.3.3.2, 6.7.2.2}. National strategies provide a mechanism to operationalize the Convention on Biological Diversity's fifteen guiding principles for the prevention, control and mitigation of impacts of invasive alien species, which remain highly relevant but are not yet adequately implemented *(well established)* {6.1.2, 6.2.3.2, 6.3.3.3, 6.7.2.3}. National strategies are central to guiding actions to implement context-specific integrated governance for addressing biological invasions *(established but incomplete)* {6.2.4, 6.7.1}.

6 An open, interoperable information platform can effectively support changing information needs on biological invasions and enable the rapid flow of information for decisions across international, national and local levels *(well established)* {6.6.2.3, 6.6.2.4, 6.6.3, 6.7.2.6}. Such a platform could ensure that knowledge is readily available to all stakeholders involved in addressing biological invasions, particularly for those whose actions are currently limited by a lack of resources *(well established)* {6.6.2.3, 6.6.3, 6.7.2.6}. Integrated data workflows and rapid data publication could considerably reduce the time lag between the establishment of evidence and making the evidence available for a wider community and for policy, including regarding Target 6 of the Kunming-Montreal Global Biodiversity Framework *(well established)* {6.6.2.3}. Conforming to the data principles of findability, accessibility, interoperability and reusability (FAIR) makes data easier to access and use for monitoring, modelling and forecasting *(established but incomplete)* {6.6.2.4}. Continuously collecting and sharing up-to-date information on biological invasions can help evaluate the effectiveness of policy instruments and management actions *(established but incomplete)* {6.6.2.3, 6.3.3} and improve management outcomes *(well established)* {6.6.2, 6.6.3, 6.7.2.3}. Such an information platform could maintain and deliver indicators of the different dimensions of biological invasions to track progress at national and global scales, and in the medium to long term as part of a responsive and adaptive policy environment {6.1.2, 6.1.3, 6.2.3, 6.6.3, 6.7.2.6} *(well established)*. Targeted investment in specific research and monitoring programmes can rapidly and effectively deliver relevant data and information for policies and management of biological invasions *(well established)* {6.6.2, 6.7.2.6}.

7 Committed engagement with stakeholders and Indigenous Peoples and local communities can benefit the management of biological invasions by improving understanding and awareness, social learning, collaboration, surveillance and data generation *(well established)* {6.2.3.3, 6.4.2.1, 6.7.2.6}. Inclusive engagement can help build policy and management plans to address biological invasions that are coherent, legitimate and reflect local environmental and cultural realities. Adaptive-collaborative governance can foster collaboration and coordination grounded in disciplinary integration, experimentation, monitoring, the use of the best available technology and social learning *(established but incomplete)* {6.2.3.3, 6.4.3}. Engagement activities can be explicitly linked with the measurement and monitoring of management actions through national strategies aimed at enhancing respect for Indigenous Peoples and local communities' knowledge, rights and priorities *(established but incomplete)* {6.4.1, 6.4.3.2}. Biocultural community protocols developed by Indigenous Peoples and local communities can frame how they wish to be engaged in the activities that impact them *(established but incomplete)* {6.4.3.2}. These protocols can facilitate a deeper engagement with the knowledge and customary governance systems of Indigenous Peoples and local communities within, rights-based frameworks and in accordance with national legislation, benefitting both good quality of life and effective management of biological invasions *(established but incomplete)* {6.4.3.2}. Social research can help to better inform management and policy and build trust between sectors of society *(established but incomplete)* {6.4.4, 6.6.1.4}. Social research can also provide valuable information on how best to share knowledge and on invasive alien species status and trends on land managed by stakeholders and Indigenous Peoples and local communities *(well established)* {6.4.1, 6.6.1.5, 6.2, 6.4}.

8 The current understanding of the biological invasion process, which includes extensive information on many of the currently most impactful species and well-established principles for prevention and control, is adequate for guiding effective action on invasive alien species *(well established)* {6.1.2, 6.1.3, 6.2.3, 6.2.4}. The complexity and uncertainty of social, economic and environmental costs and benefits of invasive alien species is broadly acknowledged *(well established)* {6.2.2} and is a central obstacle to predicting biological invasions, including the outcomes of invasion, necessitating a precautionary approach *(well established)* {6.1.2, 6.2.2, 6.3.1.2}. Nonetheless, the key aspects of this complexity are understood, including the multiple sectors and stakeholders that contribute to and are affected by invasive alien species *(well established)* {6.2.3}. Which sectors and stakeholders are involved is context dependant and their specific roles depend on the invasive alien species involved and the ecosystems affected *(well established)* {6.2.3.3, 6.3.1.1, 6.4.1, 6.7.2.5}. A further dimension of complexity that can be considered in the governance and management of biological invasions is the interaction between invasive alien species and other key drivers of change in nature, including climate change, pollution and land-use change *(well established)* {6.3.1.3}. Given the strong interactions between these drivers, considering all forms of global environmental change can yield benefits for effective environmental governance *(well established)* {6.2.3.3, 6.3.1.3, 6.7.2.2}. Despite these complexities and the uncertainty that can affect decision-making, scientific and technical solutions exist for designing efficient and effective options to deal with biological invasions {6.2.3}, and for supplying the information needed to support policy and management decisions *(well established)* {6.6.2}. Strategic investment in research to keep data and information up to date and to fill key gaps will improve the efficiency and effectiveness of management of biological invasions *(well established)* {6.6.1} **(Table 6.10)**.

9 An integrated governance approach that connects and combines key strategic interventions and creates robust governance system properties can bring about context-relevant transformative change for the effective prevention and control of invasive alien species *(established but incomplete)* {6.2.4, 6.7.1} (Figure 6.21). Positive transformation can be achieved by (a) strengthening the connectivity within the invasion governance system, using information technology and international partnerships {6.6.2.7, 6.7.2.6}; (b) developing stronger and broader global regulatory instruments to address invasions threats {6.3.1, 6.7.2.3} as well as higher visibility of biological invasions in national legislations and environmental actions plans {6.2.4.2, 6.3.2, 6.7.2.1}; (c) engaging all relevant sectors including health, environment, agriculture, fisheries and forestry, and all relevant stakeholders including the private sector, the general public and Indigenous Peoples and local communities{6.2.3.3, 6.3.1.1, 6.4.2.2, 6.7.2.4, 6.7.2.5}; and (d) supporting innovative science and environmentally sound technologies for solutions-focused approaches {6.3.3.4, 6.7.2.6}. Governance that is responsive to changes in biological invasion risk and management contexts, focuses on effective implementation, sustains investment and commitment to goals, and is equitable and inclusive across both those affected and responsible, can bring about a step change in the prevention and control of invasive alien species *(well established)* {6.2.4, 6.7.3}.

6.1 INTRODUCTION AND OUTLINE

This chapter evaluates past and possible future governance models and challenges, policy instruments and support tools, collectively called response options (**Glossary**). These response options are aimed at managing biological invasions (**Chapter 1**, **Figure 1.1**; **Glossary**) to reduce their impacts on nature, nature's contributions to people and good quality of life (**Box 6.1**; **Glossary**). Governance models (**section 6.2**) target specific components of the socioecological system, are complimentary and tend to draw on specific sets of policy instruments, tools and methods. Policy instruments (**sections 6.3** and **6.5**) are the set of options (means or mechanisms) used at any scale by individuals or organizations for building or strengthening international, national and local efforts to manage biological invasions. Policy support tools and methods (**section 6.6**) are approaches that can inform, assist and enhance relevant decisions, policy-making and implementation at the local, national, regional and international levels, to better prevent and control these species and their impacts (**Glossary**).

Multiple international organizations and programmes have highlighted the roles of governance models **(Table 6.1)**, policy instruments and policy support tools and methods **(Table 6.1)** as means to achieve international policy targets for preventing and controlling invasive alien species and managing biological invasions (**Chapter 1**, **Box 1.1**). They include the Convention on Biological Diversity (CBD), the International Union for Conservation of Nature (IUCN)

Box 6 1 **Rationale of the chapter.**

This chapter builds on previous chapters of the IPBES invasive alien species assessment to present response options for improving and strengthening the governance for biological invasions. The chapter explores governance models and policy instruments (legal, regulatory and incentive-based) and support tools available for multilateral efforts and national strategies to prevent and control invasive alien species. Together, the integration of these strategic actions could lead to positive transformative change (**Glossary**).

Guiding questions:

- What are the challenges facing biological invasions governance?
- What are the current gaps, overlaps and inconsistencies in existing legal and regulatory instruments focused on biological invasions management (**Glossary**)?
- Which decision and engagement tools can be used to manage biological invasions?
- What economic instruments can be implemented to fund or promote the various prevention, eradication, containment, mitigation, restoration and ecosystem-based management options (**Glossary**)?
- How to develop information systems (**Glossary**) to help design, implement and monitor response options to the biological invasions problem?

Keywords:
Governance, policy instruments, community engagement, integrated governance, goals and targets, multilateral coordination, implementation strategies, coherent policy regimes, open data, information systems.

Table 6 1 **Array of possible governance and policy response options for managing biological invasions.**

Possible response options that can be used to (i) prevent or mitigate the main drivers responsible for biological invasions and (ii) prevent the impacts of invasive alien species on nature and society. The response options include models of governance, categories of policy instruments and families of support tools and methods (IPBES, 2019a).

Governance Models	Categories of Policy instrument	Families of support tools and methods (main governance model/s in brackets)
A. Hierarchical	Legal and regulatory	Assembling data and knowledge (B)
B. Scientific-technical	Economic and financial	Assessment and evaluation (B)
C. Governing strategic behaviour	Rights-based and customary norms	Public discussion, involvement and participatory process (D)
D. Adaptive-collaborative	Social and cultural instruments	Selection and design of policy instruments (A, B)
		Implementation, outreach and enforcement (A, C)
		Training and capacity-building (B, D)
		Social learning and innovation (D)

Invasive Species Specialist Group (ISSG), the Food and Agriculture Organization of the United Nations (FAO), the World Health Organization (WHO), the International Plant Protection Convention (IPPC), the World Organisation for Animal Health (WOAH, founded as OIE), the World Trade Organization (WTO), the International Maritime Organization (IMO), the World Bank and the Centre for Agriculture and Biosciences International (CABI). Many of these organizations participate in the Inter-agency Liaison Group on Invasive Species established by the Executive Secretary of the CBD.

Guided by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) conceptual framework (**Chapter 1**, **section 1.6.1**; **Figure 6.1**), possible options to prevent and manage biological invasions outlined in this chapter respond to direct and indirect drivers of change responsible for biological invasions, and to the possible impacts of invasive alien species on nature, nature's contributions to people and good quality of life. **Section 6.2** presents options to strengthen the governance systems within which policy instruments and support tools for biological invasions are implemented. These include options for enhancing the coverage and strategic nature of biological invasions governance, dealing with multiple levels and sectors of society, as well as enhancing governance capabilities. **Section 6.3** evaluates the limitations of and opportunities for the current array of legal and regulatory frameworks, highlighting the need for strategies that bridge and coordinate across sectors, geopolitical units and stages of the biological invasion process. **Section 6.4** lays out the role of widespread engagement and the context and reasons for the broad and specific inclusion of stakeholders and Indigenous Peoples and local communities. This section examines the range of approaches for coordination and collaboration and shows how stakeholder and Indigenous Peoples and local communities' engagement can improve biological invasions governance. **Section 6.5** outlines financial and economic policy options that provide incentives for international organizations, governments, financial institutions and individuals to invest in invasive alien species prevention, containment, mitigation, or eradication (**Glossary**). This section assesses the role of tariffs, cost-sharing and penalty systems as deterrents to invasive alien species (**Glossary**) introduction and spread and as a financing alternative. This section also presents support tools for analysing the costs and benefits of invasive alien species. **Section 6.6** specifies options for generating and maintaining the information and knowledge needed to govern and manage biological invasions. This section identifies knowledge gaps and options for access to the information needed for the creation of early warning systems, for forecasting the spread and impact of invasive alien species and for assessing management effectiveness. Information needs for the purpose of developing and reporting on the effectiveness of policy instruments are also described. Finally, **section 6.7** summarizes the key governance and policy challenges and opportunities for sustainable biological

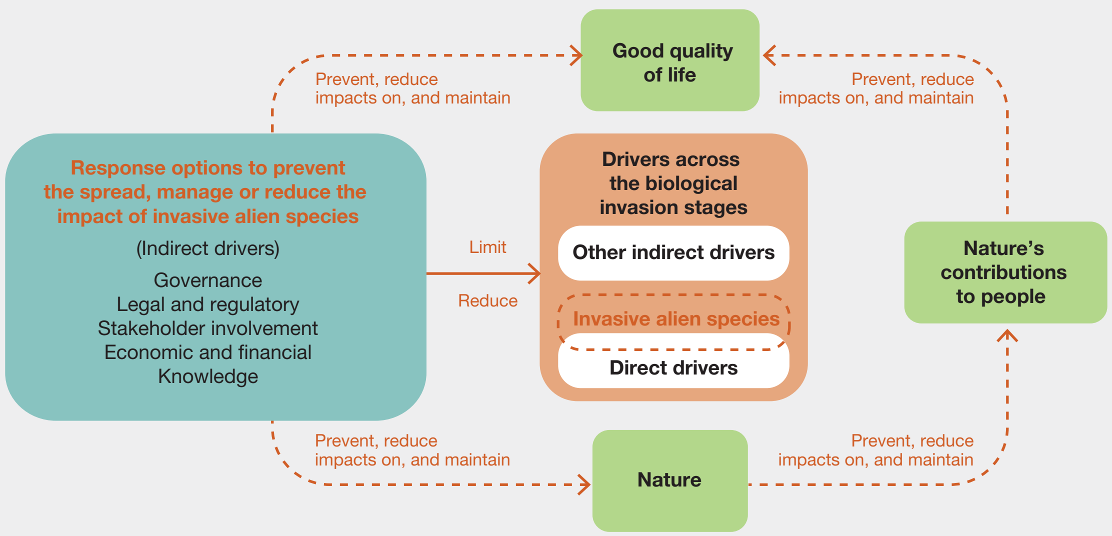

Figure 6 1 **Framework for governance and policy response options and how they relate to nature, nature's contributions to people and good quality of life.**

Simplification of the IPBES conceptual framework that describes the array of response options (left box) to limit the drivers facilitating biological invasion and the spread of invasive alien species (middle box) and mitigate their negative impacts on good quality of life (surrounding boxes connected by dashed arrows). Modified from S. Díaz *et al.* (2015), https://doi.org/10.1016/j.cosust.2014.11.002, under license CC BY-NC-SA 3.0.

invasion management that emerge from the assessment. This section outlines the set of strategic actions and governance system properties that can jointly construct an integrated approach to the governance for biological invasions that can bring about a step change in progress to achieving related policy and management goals.

6.1.1 Risks and opportunities

The IPBES invasive alien species assessment and previous IPBES assessments (as outlined in **Table 6.2**) identify risks and opportunities that provide the basis for determining future options and strategies to prevent and mitigate the impacts of invasive alien species on nature, nature's contributions to people and good quality of life **(Figure 6.1)**. These risks and opportunities can be categorized into ten groups **(Table 6.2)** and lay the foundation for the strategic identification of governance policy instruments and options. These risks and opportunities also reveal where adaptation is the most feasible option, when eradication or adequate management is no longer feasible, and when such an adaptive response could lead to a positive system transformation.

Table 6 2 **Risks (hazard or impact taking place) and opportunities (circumstances that make it possible to act) for managing biological invasions as defined in the IPBES invasive alien species assessment and other IPBES assessments.**

The risks and opportunities showcased provide an overview of the main governance challenges, policy instruments and knowledge needs that are highlighted in this chapter to effectively manage biological invasions. The relevant sections within the IPBES invasive alien species assessment and other IPBES assessments are highlighted for each of the points raised.

Risks and/or opportunity	Description (relevant chapter sections of this/previous IPBES assessments)
Information disparity	• Geographic, taxonomic, data access and publication biases (**Chapter 6**, **section 6.6.1**; **Chapter 4**, **section 4.7.2**; IPBES, 2018a) • Comparatively limited knowledge of invasive alien species impacts on fisheries, coral reefs and marine ecosystem functioning (**Glossary**; **Chapter 4**, **section 4.7.2**)
Information uncertainty	• Uncertainty about management cost, efficacy, limitations, success, collaborations and adaptation (**Chapter 5**; IPBES, 2018b) • Uncertainty about dispersal pathways, transboundary and regional collaboration (**Glossary**; **Chapter 3**, **section 3.6.1**; IPBES, 2018b) • Uncertainty about the impacts of novel assemblages/ecosystems emerging due to invasive alien species (**Chapter 4**)
Framing biological invasions in terms of their interactions with other environmental problems	• Inadequate integration of biological invasion problems in policies and management interventions aimed at addressing other environmental problems such as land degradation or climate change (**sections 6.3.1** and **6.7**; IPBES, 2016, 2018d)
Policies as drivers facilitating biological invasions	• Lack of coordination of policies on environment, infrastructure, health, agriculture, forests, environment, nutrition and biological invasion management can catalyse invasive alien species spread (**section 6.3.1**; IPBES, 2018d) • Inefficient policies on climate change, pollution, human population, land-use change, pet and wildlife trade, afforestation, horticulture and health had severe consequences on spread and impacts of invasive alien species (**section 6.3**) • Lack of coordination between policies of different sectors (**section 6.3.1**) and between national and regional regulations (**sections 6.3.2** and **6.3.3**)
Impact disparity	• Management of biological invasions is particularly difficult when they simultaneously have both serious negative environmental impacts and benefits for good quality of life or are characterized by value conflicts (**Chapters 4** and **5**) • Use of potentially invasive alien species in programmes aimed to ensure food security or promote economic development (**section 6.3**)
Technological advancement	• Absence of early detection systems and rapid response actions for prevention and eradication of invasive alien species (**section 6.6**) • Lack of monitoring (**Glossary**) programmes to track the effectiveness of responses and progress in managing biological invasions (**section 6.2**) • Lack of inter-operable and standardized databases (**section 6.6**; IPBES, 2018c, 2019a)
Economic synergies	• Conflict of interest in international regulatory frameworks with a direct or indirect focus on preventing invasive alien species introductions (**sections 6.3** and **6.5**) • Difficulties with assessing the collective investment in prevention and control of invasive alien species, how adequate they are, and costs avoided as a result (**section 6.1**)

Table 6 2

Risks and/or opportunity	Description (relevant chapter sections of this/previous IPBES assessments)
Societal response	• Lack of public support, financial resources and awareness of risks associated with invasive alien species among all stakeholders (**sections 6.4** and **6.5**, IPBES, 2018a, 2019a) • Absence of institutions that coordinate and provide oversight on invasive alien species prevention, control and mitigation strategies, and promote the flow of information (**sections 6.3** and **6.7**)
Engagement with Indigenous Peoples and local communities	• Engaging with stakeholders and Indigenous Peoples and local communities (**section 6.4**) • Lack of integration of stakeholder and Indigenous and local knowledge (**Glossary**) and limited engagement of stakeholders and Indigenous Peoples and local communities in decision-making for biological invasions (**section 6.4**)
Gaps in knowledge of governance	• Lack of information about the success of governance in management interventions (**section 6.2**) • Limited dedicated interdisciplinary research on the governance for biological invasions in an environmental governance context (**section 6.2**)

6.1.2 Progress towards international and national goals and targets for invasive alien species

The CBD is currently the most encompassing and directly relevant global environmental governance mechanism for biological invasions. It has the following three objectives: the conservation of biological diversity; the sustainable use of the components of biological diversity; and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources (**Figure 6.1**, **section 6.1.2**). The Conference of the Parties (COP) to the CBD has also specifically recognized that invasive alien species represent one of the primary threats to biodiversity, especially in geographically and evolutionary isolated ecosystems, such as small island developing States (SIDS). As early as 2002, the CBD COP adopted a series of Guiding Principles for improving the governance for biological invasions **(Table 6.3)**. These Guiding Principles remain highly relevant and provide a list of options to accelerate and sustain progress on invasive alien species and their control. The responsibilities of Parties to the CBD are therefore longstanding, and gaps and shortcomings in the governance for biological invasions over the last several decades (**section 6.1.3**) have meant that actions have not been sufficient to stop their spread (**Chapter 2**).

Under the recent Kunming-Montreal Global Biodiversity Framework (CBD, 2022a), the invasive alien species target (Target 6) encompasses eliminating, reducing and mitigating impacts through pathway management (**Glossary**), prevention and with a focus on priority species and priority sites (**Chapter 1**, **Box 1.1**). With the addition of some new elements Target 6 reinforces the key elements of the previous Aichi Biodiversity Target 9, based on which current progress has been evaluated. Assessment of the progress towards meeting Aichi Biodiversity Target 9 (invasive alien species prevented and controlled) concluded that, while increases in the adoption of related policy was encouraging, there was still a considerable gap between the development and adoption of invasive alien species policy, and implementation at national levels (Secretariat of the CBD, 2020; **Table 6.4**).

The fifth edition of the Global Biodiversity Outlook also indicated that there has been no reduction in the pressure from invasive alien species on biodiversity, ecosystems and society (Secretariat of the CBD, 2020). Factors identified as underlying the imperfect achievement of Aichi Biodiversity Target 9 included: inadequate policy implementation due to limited capacity and resourcing of relevant governmental agencies; lack of coherence across multiple, relevant policies; and the fact that policy adoption does not equate directly to management effectiveness. This Global Biodiversity Outlook report further pointed out the lack of research and data on biological invasion policy effectiveness at a global scale (Secretariat of the CBD, 2020). Subsequent studies have also identified poor governance as a factor limiting national-level progress to achieving other Aichi Biodiversity Targets (Buchanan *et al.,* 2020).

The Sustainable Development Goals (SDGs) also include a target for invasive alien species (Target 15.8; **Chapter 1**, **Box 1.1**) closely related to Aichi Target 9 of the Strategic Plan for Biodiversity 2011-2020. This target aims to track progress in the commitment by countries to relevant multilateral agreements, and the proportion of countries with national strategies, legislation and policy for invasive alien species. However, the indicators that were selected to track progress of Target 15.8 of the 2030 Agenda for Sustainable Development also included for the first time an “input response” element (i.e., the extent to which the measures identified are resourced; **section 6.2.1**).

Table 6 3 **The 15 Guiding Principles for the prevention, introduction and mitigation of impacts of alien species that threaten ecosystems, habitats or species.**

The text associated with these principles is not provided in full here – abbreviated notes are shown where particularly relevant to invasive alien species governance as discussed in **section 6.2** (**Chapter 1**, **section 1.3.1**). Source: CBD (2002).

No.	Guiding Principle
A. GENERAL	
1	**Precautionary approach:** efforts to identify and prevent unintentional introductions as well as decisions concerning intentional introductions should be based on the precautionary approach as described in Principle 15 of the 1992 Rio Declaration.
2	**Three-stage hierarchical approach:** prevention is the top priority; followed by early detection, rapid response and eradication; and then containment, long-term control measures and examination of the benefits and costs.
3	**Ecosystem approach:** as described in decision V/6 of the Conference of the Parties.
4	**The role of States:** States should recognize the risk that activities within their jurisdiction or control may pose to other states and should take appropriate individual and cooperative actions to minimize that risk, including making information on the identity of invasive alien species available to other states.
5	**Research and monitoring:** Research on an invasive alien species should focus on the history and ecology of invasions, the biological characteristics of the invasive alien species, and the associated impacts. Monitoring should involve multiple sectors and include both targeted and general surveys.
6	**Education and public awareness:** Promote education and public awareness of the causes of invasion and the risks associated with the introduction of alien species.
B. PREVENTION	
7	**Border control and quarantine measures:** Putting in place appropriate measures to control introductions of invasive alien species based on risk analysis of threats and potential pathways of entry.
8	**Exchange of information:** CBD Parties should assist in developing an inventory and synthesis of relevant databases, and developing information systems and an interoperable distributed network of databases for compilation and dissemination of information on alien species. The Parties should provide all relevant information on their specific import requirements for alien species and make this information available to other States.
9	**Cooperation, including capacity-building:** Cooperation should be based on programmes developed to share information as well as cooperative research and funding efforts. Capacity-building may involve technology transfer and the development of training programmes, especially for countries that lack expertise and resources.
C. INTRODUCTION OF SPECIES	
10	**Intentional introduction:** No first-time intentional introduction or subsequent introductions of an alien species already invasive or potentially invasive within a country should take place without prior authorization from a competent authority of the recipient state(s).
11	**Unintentional introductions:** Provisions to address unintentional introductions need to be set in place.
D. MITIGATION OF IMPACTS	
12-15	(full text not provided here) Including mitigation of impacts (no.12), eradication (no.13), containment (no.14) and control (no.15) (**Chapter 5**).

By comparison, multilateral instruments or organizations such as the IPPC and WOAH have been widely successful in developing and implementing instruments to mitigate the risks of invasive alien species considered to be pests or to affect animal health. There are over 40 adopted international standards for phytosanitary measures (ISPMs, developed by IPPC),[2] 31 Diagnostic Protocols, and 39 Phytosanitary Treatments aimed to protect the environment, forests and biodiversity while also facilitating economic and trade development. The standards and codes developed by the IPPC and WOAH have provided a foundation for multilateral collaboration in managing the risks posed by invasive alien species.

2. https://www.ippc.int/en/core-activities/standards-setting/ispms/

Table 6 4 **Progress against invasive alien species policy goals and targets.**

Indicator category	Indicator[3]	Elements of Aichi Biodiversity Target (AT) 9 and SDG Target 15.8	Global progress against Target or Goal
Driver	Trends (**Glossary**) in pathways of introduction and spread	Measures are in place to manage pathways to prevent introduction and establishment (AT); Introduce measures to prevent the introduction of invasive alien species (SDG)	Progress has been made, but at an insufficient rate. This target has not been achieved (high confidence; (Secretariat of the CBD, 2014, 2020). Major pathways are not efficiently controlled at a global scale (Secretariat of the CBD, 2014), but major advancements have been made in the context of shipping (in particular, an agreement to prevent biological invasions *via* ballast water).
		Pathways identified and prioritized (AT 9.2)	Major pathways have been identified (Faulkner *et al.,* 2020; IUCN, 2017; Saul *et al.,* 2017; Secretariat of the CBD, 2014). However, the pathways of introduction of more than a third of introduction events are unknown (McGrannachan *et al.,* 2021).
Pressure	Trends in numbers of invasive alien species and their impacts	None	The number of documented, new introductions of alien species continues to increase (Seebens *et al.* 2017). Progress towards target has been made, but at an insufficient rate (Secretariat of the CBD, 2014).
		Invasive alien species are identified and prioritized (AT 9.1)	Measures have been taken in many countries to develop checklists of invasive alien species (Secretariat of the CBD, 2014, 2020). Target partially achieved.
State	Trends, mechanisms and severity of invasive alien species impacts	Introduce measures to significantly reduce the impact of invasive alien species (SDG)	A negative trend in the conservation status of species threatened by invasive alien species in the Red List Index (McGeoch *et al.,* 2010) suggests that this target has not been achieved. Overall, there has not been an improvement in conservation status of species threatened by invasive alien species, although some progress has been made for some species and for species on islands (CBD, 2020b; Secretariat of the CBD, 2014).
Response Input	Trends in the allocation of resources towards the prevention or control of invasive alien species	Proportion of countries adopting relevant national legislation and adequately resourcing the prevention or control of invasive alien species (SDG)	Of the 195 countries party to the CBD, almost half have no national budget and no funding *via* global mechanisms for invasive alien species prevention and control activities (Pagad *et al.,* 2020).
Process	Trends in establishment and national adoption of international agreements relevant to the prevention and control of invasive alien species	Trends in policy responses, legislation and management plans to control and prevent spread of invasive alien species (Pagad *et al.* 2020)	Between 30 and 90 per cent of all countries are signatory to the nine multilateral agreements relevant to the prevention or control of invasive alien species, including the CBD, with most countries' signatory to the World Heritage Convention, IPPC and the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES; 2% increase since 2010; Pagad *et al.,* 2020). Likewise, most of the world shipping tonnage (over 91%) is under regulation under the International Convention for the Control and Management of Ship's Ballast Water and Sediments (BWM Convention; IMO, 2022). Trends in adoption overall are positive since 1970 (Pagad *et al.,* 2020).
	Trends in numbers of countries with national legislation and other policy measures relevant to the prevention and/or control of invasive alien species	Proportion of countries adopting relevant national legislation and adequately resourcing the prevention or control of invasive alien species (SDG)	Only 10% of reporting parties have national targets of similar scope and ambition to Aichi Biodiversity Target 9 and are on track to meet them (CBD, 2020b). Party self-assessment is variable as assessed against their own national targets (Secretariat of the CBD, 2020). Most countries (190) party to the CBD have some form of national legislation relevant to invasive alien species; 17% of these are specifically focussed on invasive alien species (Pagad *et al.,* 2020). 39% of countries have developed a national invasive species strategy and action plan (NISSAP; Pagad *et al.,* 2020). 10% of countries rely entirely on international funding for invasive alien species prevention and control activities (Pagad *et al.,* 2020). Of the countries party to the CBD (195), 80% have invasive alien species targets in their national biodiversity strategies and action plans (NBSAPs), 74% are aligned with Aichi Biodiversity Target 9 (Pagad *et al.,* 2020).
Output	Trends in the prevention of invasive alien species	Measures are in place to manage pathways to prevent introduction and establishment; Priority species are controlled or eradicated (AT 9.4); Introduce measures to control or eradicate priority invasive alien species (SDG)	There has been no significant overall progress towards this target (Secretariat of the CBD, 2014). Some measures have been put in place but are not sufficient to prevent the continuing increase in invasive alien species.

Table 6 4

Indicator category	Indicator[3]	Elements of Aichi Biodiversity Target (AT) 9 and SDG Target 15.8	Global progress against Target or Goal
Output	Growth in information relevant to informing policy on invasive alien species prevention and control	none	In progress
Outcome	Trends in successful control and eradications of invasive alien species	Priority species are eradicated (AT 9.3)	Progress towards target, but at an insufficient rate (Secretariat of the CBD, 2020). Some control and eradication, but data limited. Progress has been made, but Target has not been achieved.25% of invasive alien species mammal eradications on islands have occurred since 2010 (Secretariat of the CBD, 2014, 2020).
		Priority species are controlled (AT 9.3)	Data limited (Secretariat of the CBD, 2020). Target unlikely to have been achieved.

3. The indicator here is expressed in an inclusive general form, encompassing relevant alternative formulations of closely related indicators.

6.1.3 Specific progress towards governance-related invasive alien species goals

Response plans and monitoring for invasive alien species

In general, goals specific to societal "responses" to biological invasions are very poorly developed, as is the availability of data on response plans and response monitoring (**section 6.6.3**; Vicente *et al.,* 2021). Nonetheless, some components of existing invasion targets and associated indicators fall into this "response" category, and these are highly relevant to the governance for biological invasions at global and national scales. These include the existence and uptake of multilateral agreements and national legislation relevant to the prevention and control of invasive alien species, and resourcing of invasive alien species prevention and control activities (McGeoch *et al.,* 2010; Pagad *et al.,* 2020; **section 6.6.3**).

Multilateral agreements

Monitoring of the response targets that do exist (**section 6.6.3**) shows that there has been a small increase in the number of countries that are signatories of seven relevant multilateral agreements in the last decade, and country adoption ranges from about 60 per cent to 98 per cent across these agreements (**Figure 6.2**; Pagad *et al.,* 2020). Of these, IPPC and WOAH have been critical instruments for preventing the introduction of invasive alien species and defining the roles of authorities working on biosecurity (**Glossary**) to prevent introductions of invasive alien species. Likewise, the eighth, and most recent, agreement – the BWM Convention – has reached a country signatory level of 33 per cent since it was established in 2004 (**Figure 6.2**; **Chapter 5**, **section 5.5.1**), although it was ratified only in 2017. The over 60 country signatories to this convention are responsible for 91 per cent of the world's shipping tonnage (IMO, 2022), making it a potentially powerful instrument for preventing invasive alien species; interestingly, this is the only multilateral treaty adopted specifically to prevent the spread of invasive alien species.

National legislation

The development and adoption of relevant national legislation is split across agricultural and environmental sectors, and in some cases also split across industries involving plants and those involving animals **(Figure 6.3)**. Only 17 per cent of countries have invasive alien species-dedicated national legislation (Pagad *et al.,* 2020), whereas an estimated 69 per cent have invasive alien species-specific legislation as part of legislation in other sectors (in addition to plant and animal health legislation that is broadly relevant to invasive alien species; Pagad *et al.,* 2020).

Overview of progress

The development of action plans for invasive alien species by CBD parties can help them achieve biological invasion-relevant goals and targets, such as occurred under the previous Aichi Target 9 of the 2011-2010 Strategic Plan (CBD, 2020). However, across the elements of Aichi Biodiversity Target 9 and related Target 15.8 of the 2030 Agenda for Sustainable Development, limited progress was made over the decade to successfully prevent, control

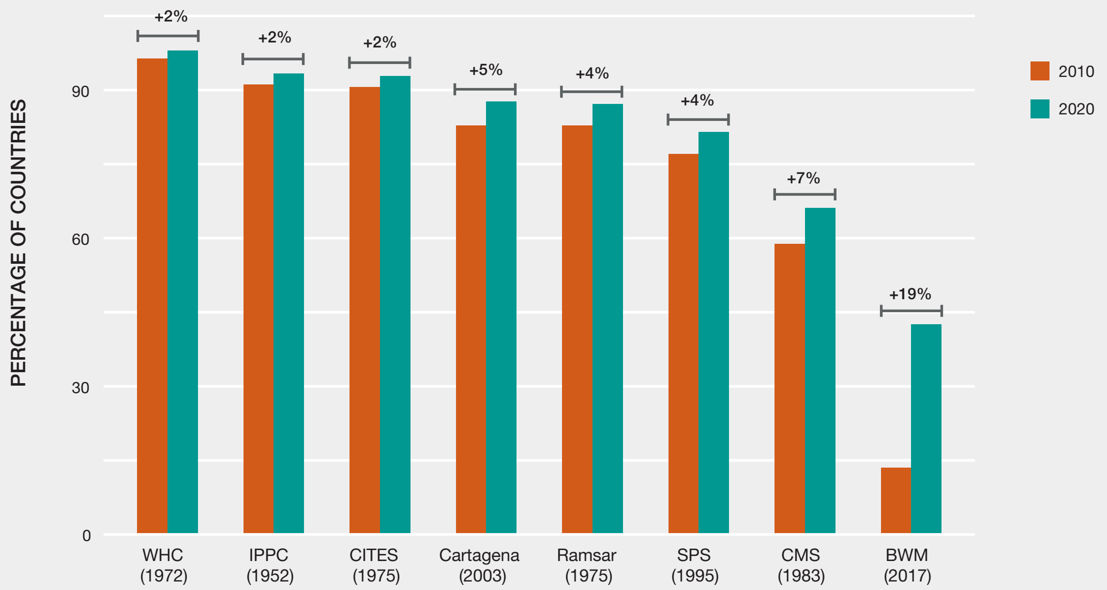

Figure 6 2 **Percentage of countries (y axis) signatory to eight multilateral agreements relevant to the prevention and control of invasive alien species (x axis).**

Data is shown for 2010 (left bars; n = 192) and 2020 (right bars; n = 195), with % increase since 2010 (shown above), signatory countries to eight multilateral agreements relevant to the prevention and control of invasive alien species. Only countries party to the CBD at the time of reporting were considered in the analysis. The eight multilateral agreements (year of establishment below acronym in figure) analysed were the Cartagena Protocol on Biosafety to the CBD (CBD, 2000), the IPPC (IPPC, 1952), the Agreement on the Application of Sanitary and Phytosanitary Measures of the WTO (WTO SPS; WTO, 1995), CITES (CITES, 1975), the Convention on Wetlands of International Importance (Ramsar) (Secretariat of Convention on Wetlands, 1971), the Convention on the Conservation of Migratory Species of Wild Animals (CMS) (CMS, 1979), the World Heritage Convention (WHC) (UNESCO, 2017) and the BWM Convention) (IMO, 2004). The WOAH (WOAH, 2011) has a high level of uptake (93%) and showed no change between 2010 and 2020 and is therefore not included in the figure. Source: Pagad *et al.* (2020), https://opal.latrobe.edu.au/articles/report/International_Adoption_of_Invasive_Alien_Species_Policy/13065158, under license CC BY 4.0.

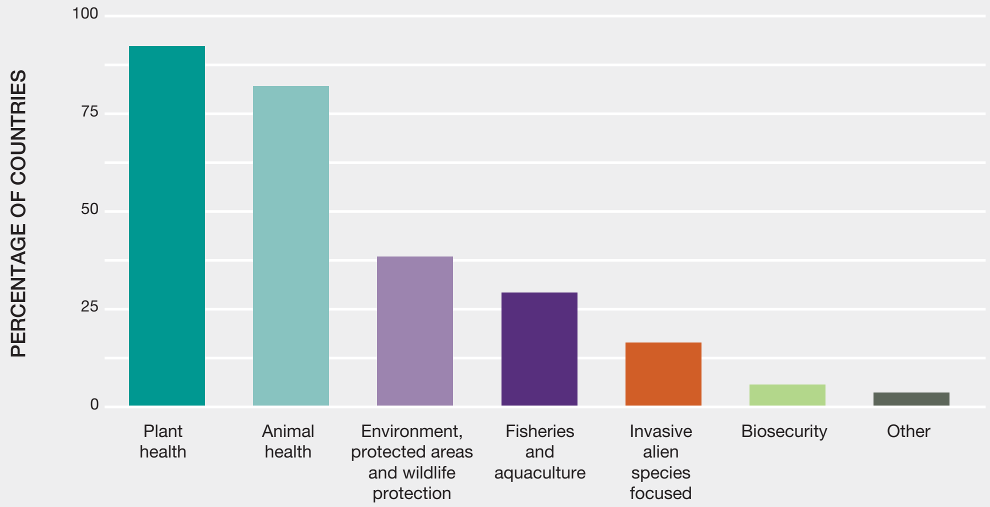

Figure 6 3 **Adoption of national legislation relevant to the prevention and/or control of invasive alien species.**

Data is shown for 195 countries reporting to the CBD. The percentage of countries (y axis) with national legislation in invasive alien species-relevant sectors (x axis) shown. Source: Pagad *et al.* (2020), https://opal.latrobe.edu.au/articles/report/International_Adoption_of_Invasive_Alien_Species_Policy/13065158, under license CC BY 4.0.

and reduce the impacts of invasive alien species (**section 6.1.2**; **Table 6.4**). Most countries (about 196 countries; Pagad *et al.,* 2022) now have checklists of introduced and invasive alien species, but the documented numbers of new introductions continue to increase (**Chapter 2**; Seebens *et al.,* 2017). The overall conservation status of species threatened by invasive alien species (Blackburn *et al.,* 2019) continues to worsen, many countries have little to no funding for invasive alien species activities (Blackburn *et al.,* 2019; **section 6.1.2**), and most national invasive alien species targets lacked ambition relative to Aichi Target 9 **(Table 6.4)**. Legislative and other policy instruments for invasive alien species are highly variable across countries and across sectors within countries (Pagad *et al.,* 2020; **section 6.1.2**). Data available for assessing the management of pathways of introduction of invasive alien species and of alien species (**Chapter 5**, **section 5.3.1**), and the effectiveness of this management, are inadequate and largely unavailable **(Table 6.4)**.

Resourcing

Estimates of the financial cost of biological invasions to countries vary widely, depending on the data source, location and evaluation method used (Diagne *et al.,* 2020). Based on country-sourced data (Pagad *et al.,* 2020), estimates of country investment in the prevention and control of invasive alien species (**section 6.5**) show that close to half of countries allocate no funds, with most such countries concentrated in Africa (**Figure 6.4** and **Figure 6.5**). Indeed, Africa depends most heavily on globally-sourced funding for the prevention and control of invasive alien species **(Figure 6.5)** and needs additional resources to support policy development and reporting (Egoh *et al.,* 2020). In the other IPBES regions, funds are allocated through a mix of national and international sources **(Figure 6.5)**. Europe and Central Asia have the highest rates of nationally derived funding. However, even where relevant legislation has been adopted, countries face significant resource shortages (Outhwaite, 2018). It is important to note that in these indicators, investment in resourcing of biological invasions policy and management implementation is different from the realized "cost" of invasive alien species measured as damage or loss from invasive alien species and expenditure on management (Diagne *et al.,* 2020).

In summary, globally representative, country-relevant data on the governance for biological invasions and related policy instruments show generally high levels of compliance with multilateral agreements. Such agreements do contribute to prevention and control of invasive alien species and stimulate the existence of many different national instruments, though in most countries these

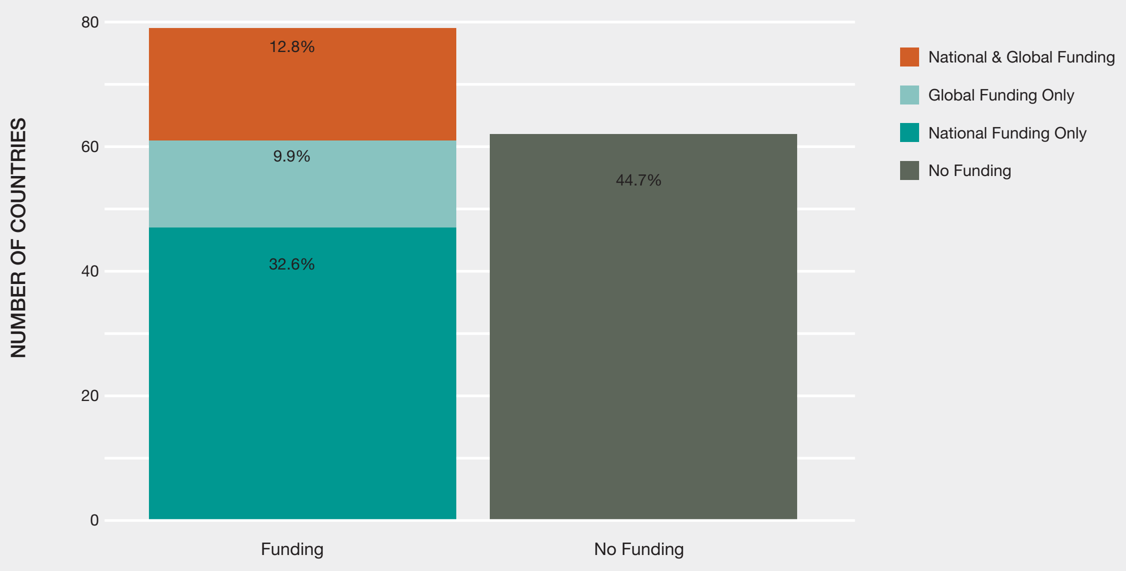

Figure 6 4 **Allocation of financial resources to the prevention and control of invasive alien species, including both national and global financial mechanisms.**

Source: Pagad *et al.* (2020), https://opal.latrobe.edu.au/articles/report/International_Adoption_of_Invasive_Alien_Species_Policy/13065158, under license CC BY 4.0.

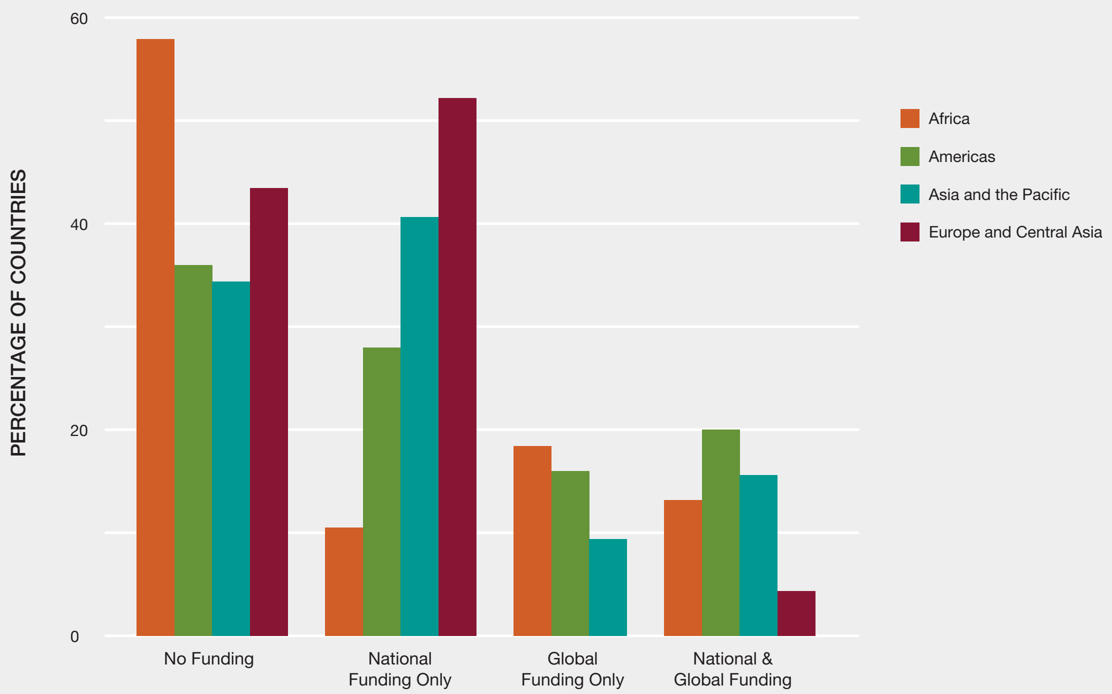

Figure 6 5 **Percentage of countries (y axis) per IPBES region that have accessed different funding sources (x axis) for invasive alien species prevention and control.**

Africa: n = 38, Americas: n = 25, Asia and the Pacific: n = 32, Europe and Central Asia: n = 46. Source: Pagad *et al.* (2020), https://opal.latrobe.edu.au/articles/report/International_Adoption_of_Invasive_Alien_Species_Policy/13065158, under license CC BY 4.0.

instruments are siloed within industry sectors **(Figure 6.3)**, most of which are not dedicated to invasive alien species prevention and control. A thorough analysis of national legislative instruments for biological invasions is still lacking (perpetuating the situation identified mid-term in the Strategic Plan for Biodiversity 2011-2020), as are assessments of the effectiveness of invasive alien species legislation (Leadley *et al.,* 2014).

Although evidence for progress in establishing and advancing effective governance for biological invasions is patchy and incomplete, the weight of evidence points to a failure to adequately resource, prevent and control invasive alien species (throughout this assessment and **section 6.6.3**). While it is not possible to establish how much worse the situation would be in the absence of the substantial collective investment made to date to prevent and control invasive alien species (**Chapter 5**, **section 5.5.7**), it can be inferred that governance approaches and governance systems for biological invasions have been inadequate. Strengthening related governance provides an overarching option for improving the prevention and control of invasive alien species and making faster progress toward achieving multilateral goals and targets (Buchanan *et al.,* 2020).

6.2 GOVERNANCE RESPONSE OPTIONS

This section provides context for assessing governance options for biological invasions. It aims to clarify the understanding of the governance system within which a policy instrument or policy support tool for biological invasions is being implemented **(Box 6.2)**. "Good governance is an enabling condition for policy implementation, distributing the resulting positive impacts evenly across society" (IPBES, 2019b). This section identifies governance considerations from invasive alien species-specific literature and contextualizes these within environmental governance more broadly. A key finding is that there is little dedicated interdisciplinary research on the governance for biological invasions in an environmental governance context.

The concept of governance is defined and used in several ways. This assessment uses the formulation and rationale of Gilek *et al.* (2016):

> "Governance includes both structures – such as policy contexts, existing power relations among key actors,

Box 6 2 **Governance for biological invasions.**

Governance encompasses the norms, rules, laws, values, expectations, relationships and structures that affect or guide the behaviour of individuals and institutions, public and private. In the context of biological invasions, governance is aimed at the specific public purpose of preventing and reducing the spread and preventing the harm, caused by invasive alien species (Andonova & Mitchell, 2010; M. S. Reed & Curzon, 2015). The governance for biological invasions therefore encompasses formalized arrangements such as national strategy and legislation, as well as informal decision-making processes involving the range of effecting and affected stakeholders (**section 1.5.1** in **Chapter 1** for more information on stakeholder groups, Reed & Curzon, 2015). A key feature of governance for biological invasions is that it is a continuous, cooperative process that accommodates diverse and conflicting interests to enable action (Riley, 2012).

> regulatory frameworks and organizational forms of decision-making, reflexivity and participation – and processes. Processes comprise aspects such as the evolution of institutions and interactions between, for example, science and policy, as well as communication and interaction among policymakers, scientists, and other stakeholders. Processes also include the development of strategies, framings, communication, and learning."

Strong governance can help to address the problem of invasive alien species, as it enables the legislation, regulations, cooperation, participation and monitoring of actions to mitigate key drivers (T. Evans *et al.,* 2018).

6.2.1 The theory of change and indicator frameworks for improving implementation

The Driver-Pressure-State-Response (DPSR) model, sometimes including impacts (DPSIR; OECD, 2003), is a strategic framework used for reporting on global and national progress toward meeting goals and targets. It is designed to directly link monitoring of the problem with the actions taken to deal with it. The Kunming-Montreal Global Biodiversity Framework (CBD, 2022a) now extends this, using a theory of change for accelerating action to achieve biodiversity goals for the planet and people (CBD, 2021a); it distinguishes four types of response, i.e., input, process, output and outcome (OECD, 2019; **Table 6.5**). This more detailed and specific identification of the types of responses

Table 6 5 **Four types of societal responses that can be measured and monitored for the purpose of limiting the spread and reducing the impacts of invasive alien species.**

The four types of responses are in the context of a DPSR framework. Adapted from OECD (2019), with the addition of invasive alien species-specific examples.

Response type	Definition	Invasive alien species examples
Input	Measures the material and immaterial pre-conditions and resources – both human and financial – provided for an activity, project, programme, or intervention	• Budget allocated for invasive alien species research, education, monitoring, prevention and control • Number of staff allocated to invasive alien species monitoring, prevention and control
Process	Measures the progress of processes or actions that use inputs and the ways in which programme services and goods are provided	• A national inter-Ministerial Committee for biosecurity established • Targeted education programmes for local communities affected by invasive alien species
Output	Measures the quantity, quality and efficiency of production of goods or services because of an activity, project, programme or intervention	• New legal or policy instruments • Studies such as national invasive alien species assessments completed • The costs of invasive alien species integrated into national accounts
Outcome	Measures the intermediate broader results achieved through the provision of outputs	• Increased eradications of invasive alien species • Reduced ranges of priority invasive alien species • Reduced impacts of invasive alien species

Box 6 3 The Driver-Pressure-State-Response framework for invasive alien species.

This framework **(Figure 6.6)** is intended to guide investment in monitoring and to enable evidence-based causality to be assigned to the relationships among drivers affecting biological invasions (**Chapter 3**, **section 3.5**), invasive alien species and their impacts (**Chapter 4**) and societal responses to dealing with the problem (**Chapter 5**). The indicators listed under each part of the framework below are examples from the application of this framework to the Antarctic (for a global example see McGeoch *et al.,* 2010). For example, trends in invasive alien species eradication at different scales (response) lead to reduced numbers of alien and invasive alien species (pressure) and reduced extinction risk of species threatened by invasive alien species (state). Trends in the number of tourists in the region (driver – note that the term driver in this context differs from its general use in this assessment) provide the information needed to inform policy (response).

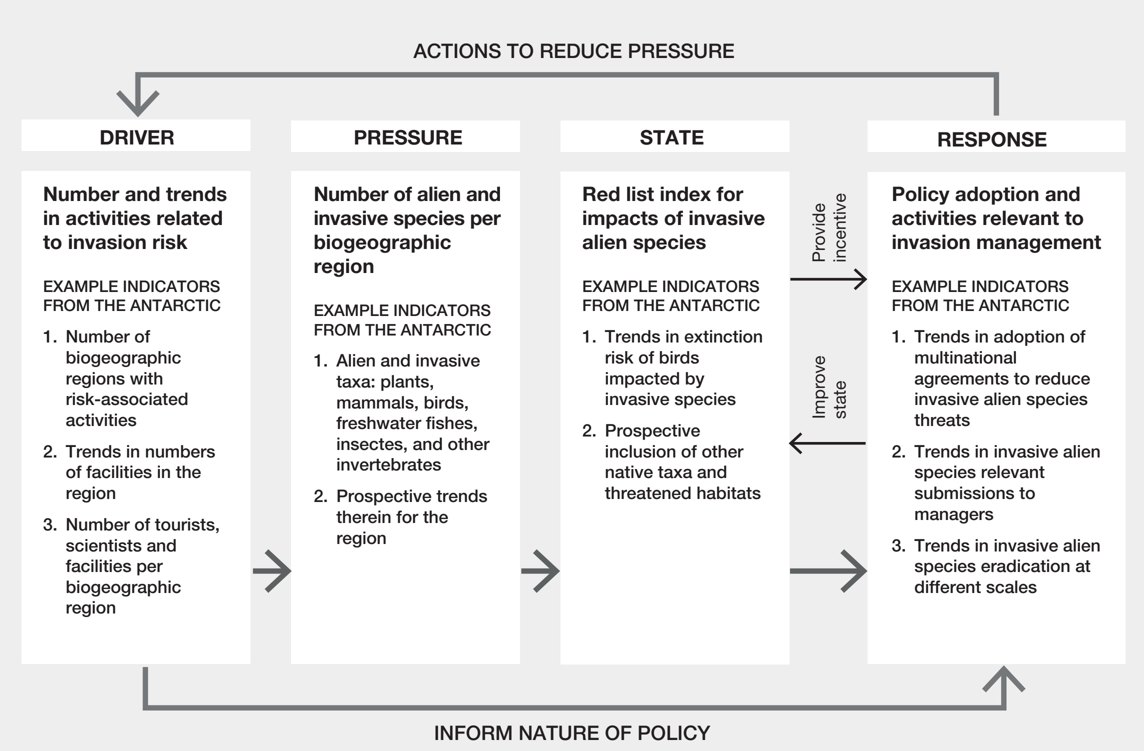

Figure 6 6 Driver-Pressure-State-Response (DPSR) framework for invasive alien species in the Antarctic context.

The DPSR applied to the Antarctic context as an example (non-bold text). Adapted from McGeoch *et al.* (2010, 2015), https://doi.org/10.1111/j.1472-4642.2009.00633.x, https://doi.org/10.1016/j.gloenvcha.2014.12.012, under license CC BY 4.0 and Elsevier license 5406200304811 respectively.

needed to bring about urgent positive change is intended to strengthen the power of monitoring, analysis and reporting (**section 6.6.2**; CBD, 2021b). In other words, if these four responses are effective, they would result in a reduction in invasive alien species pressure and an improvement in the state of socioecological systems negatively impacted by invasive alien species (**section 6.6.2**; Essl *et al.,* 2020).

The DPSR and theory of change frameworks are valuable strategic governance tools for biological invasions because, by design (OECD, 2019), they explicitly connect the causes (drivers) facilitating biological invasions, the size of the problem (pressure), its impact (state) and societal responses to dealing with it (McGeoch *et al.,* 2010; **Box 6.3**), although there are currently gaps in its application and implementation (Vicente *et al.,* 2022). By tracking change in each of these components (for example, using indicators), it becomes possible to design evidence-based, well-motivated and targeted policies for invasive alien species. The framework further makes it clear that the type, size and effectiveness of societal responses will determine the extent to which drivers decline (for example McGeoch *et al.,* 2015; **Box 6.3**). Importantly, the focus of the Kunming-Montreal Global Biodiversity Framework is on the "R, response" in

DPSR, *via* a theory of change, so that the slow progress of implementation can be accelerated.

6.2.2 Identifying the challenges of governing biological invasions

Environmental governance involves increasingly complex and interconnected arrangements, and the governance for biological invasions is no exception (Andonova & Mitchell, 2010; Gilek & Karlsson, 2016). Despite notable successes (**Chapter 5**, **section 5.5**), there are shortcomings in the prevention and control of invasive alien species, leading to a sustained global presence of invasive alien species introductions (**Chapter 2**, **section 2.2**). Understanding the underlying reasons for governance and management failures, across multiple environments, regions and taxonomic groups of invasive alien species helps to design better response options. Ten features emerged from a review of the governance challenges posed by invasive alien species.[4] These challenges are often interdependent (one may drive another for example) and jointly undermine effective prevention and control efforts (Jacobs, 2017; Linke *et al.,* 2016; J. Reed *et al.,* 2016). These key challenges are outlined below as the foundation for the options discussed in **section 6.2.4** and in the rest of this chapter.

(1) Complexity

The governance for biological invasions is considered to be complex because the process of biological invasions is naturally dynamic in space and time (**Chapter 1**, **section 1.4**). It has multiple stages and drivers, involves a large and diverse set of stakeholders, and crosses jurisdictional boundaries (Brenton-Rule *et al.,* 2016; Liu *et al.,* 2018; **Figure 6.7**; **Chapter 1**, **section 1.5.1**). The dynamic and difficult-to-predict behaviour of new technological options (such as the potential use of gene drives; **Chapter 5**, **section 5.4.4.2.j)** adds another level of complexity (Mitchell *et al.,* 2018). Context-specific application of integrated governance for biological invasions (**Glossary**) thus involves multiple trade-offs and the consideration of social, technological and ecological contexts and risks (Lubell *et al.,* 2017) across all levels of governance (Lansink *et al.,* 2018; Riley, 2012; **Figure 6.7**).

(2) Uncertainty

A high degree of uncertainty is associated with the biological invasion process because many species are involved and the likelihood of any species invading is determined by a combination of multiple biological, driver and pathway characteristics (Cooney & Lang, 2007; Udovyk & Gilek, 2013; **Chapter 1**; **Chapter 3**; **Chapter 5**, **section 5.2.2.3**; **Figure 6.7**). Biological invasion processes are non-linear and the uncertainty is “inherent, fundamental and persistent” (D. C. Cook *et al.,* 2010; Cooney & Lang, 2007). The outcomes of this complexity are difficult to predict within specific, narrow contexts, and therefore understanding the likely success of interventions is also difficult (Moon *et al.,* 2017; Smolarz *et al.,* 2016). Time lags (**Glossary**) between different parts of the invasion process, and in policy and management responses to invasive alien species, add to this uncertainty (**Chapter 1**, **section 1.4.4**; **Chapter 2**, **section 2.2**; Jacobs, 2017; J. Reed *et al.,* 2016).

(3) Information availability, flow and access

Information silos across stakeholder groups hinder effective governance (Collins, 2018; Nourani *et al.,* 2018; Peltzer *et al.,* 2019). Effective, sustained communication and collaboration across large, multi-layered networks is however difficult to achieve and has a high transaction cost (Lubell *et al.,* 2017; Nourani *et al.,* 2019). The flow of information relevant to biosecurity within and across countries and trading partners is also limited (D. C. Cook *et al.,* 2010). Actors involved at different scales and levels of governance tend to have access to different types of knowledge (Omondiagbe *et al.,* 2017). This includes the gap between science and practice (Aslan *et al.,* 2009; Esler *et al.,* 2010). There is also an imbalance of information between those who bear the costs of invasive alien species (affected actors; those who tend to have good knowledge of invasive alien species), and the actors responsible for exacerbating biological invasion risk (causal actors; **Glossary**; **section 6.6**; Cook *et al.,* 2010).

(4) Over-reliance on hierarchical governance

The currently dominant, hierarchical forms of governance for biological invasions tend to be centralized, top-down, process-heavy and reactive and, while necessary, are on their own not adequate for preventing and controlling invasive alien species (Cook *et al.,* 2010; Evans *et al.,* 2018; Reed & Curzon, 2015; **Figure 6.7**). Policy models can rely too heavily on rigid, non-adaptive, top-down approaches (Cooney & Lang, 2007). Hierarchical governance can be slow, culturally inappropriate, and not in step with the latest technological developments or scientific understanding (Barnhill-Dilling *et al.,* 2019; Boström *et al.,* 2016; Head & Atchison, 2015; Hughes & Convey, 2014; Trump *et al.,* 2018). Invasive alien species differ in key ways from other drivers of change in nature; for example, a strong precautionary approach (**Glossary**) that is often not enabled by traditional governance approaches is crucial (T. Evans *et al.,* 2018; Smolarz *et al.,* 2016). In addition, the power imbalances that can develop under highly centralized governance can lead to, for example, incoherent policy,

4. Data management report available at: https://zenodo.org/doi/10.5281/zenodo.5762739

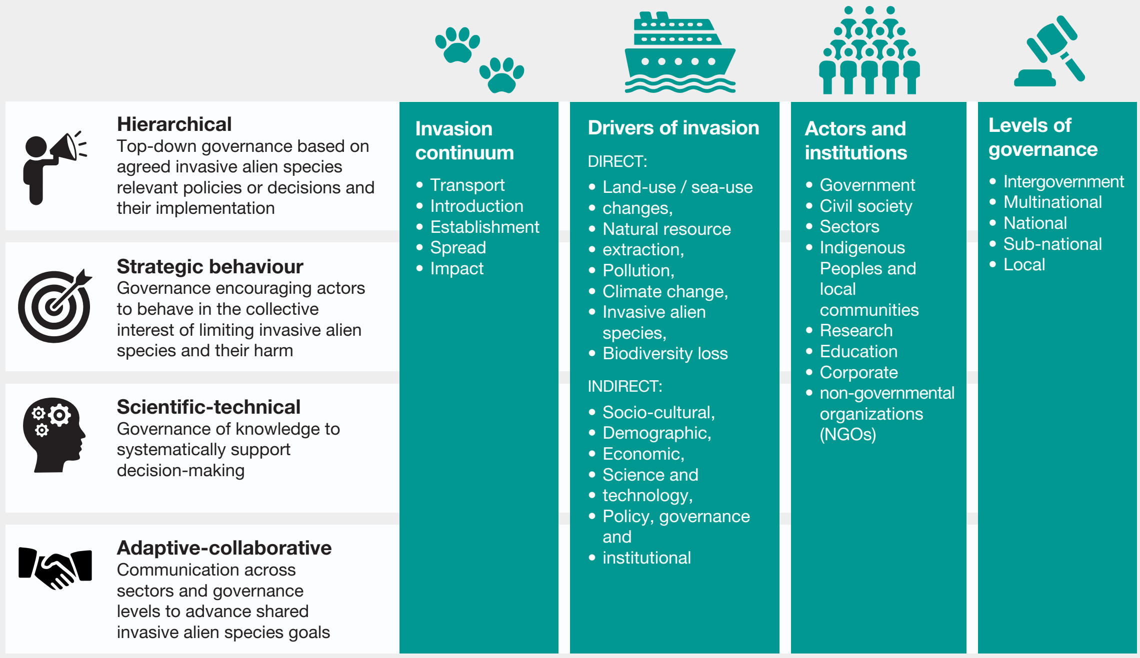

Figure 6 7 **The complex socioecological system characterizing biological invasions.**

The complex socioecological system (vertical boxes) that characterizes biological invasions and their management, and the four, complementary governance models (horizontal boxes) needed to enable successful policy implementation through integration within and across these components of the system (**section 6.2.4**). Data from: IPBES-PEG (2019b).

disengagement, or conflict amongst the broad range of stakeholders affected by invasive alien species (Neale & Macdonald, 2019; A. L. Smith *et al.,* 2013). It is now widely recognized that governments as decision-making authorities are necessary but insufficient for effective invasive alien species prevention and control (Miyanaga & Nakai, 2021; **section 6.2.3.1**).

(5) Fragmentation of policy instruments and their application

Current policy on invasive alien species and its implementation is often fragmented, with multiple, often isolated decision-making centres. As a result current policy is less effective than it could be (Gilna *et al.,* 2014; Nourani *et al.,* 2019; Praseeda Sanu & Newport, 2010; Rudd *et al.,* 2018). At the highest level, as assessed by Outhwaite (2018), there is no "full and coherent applicable body of international law". This fragmentation also includes policy differences between levels of governance and between actors and institutions, e.g., across industry sectors such as agriculture, forestry and the environment **(Figure 6.7)**, between countries and regions, and national and subnational levels of governance (Lubell *et al.,* 2017; P. Martin *et al.,* 2016). Fragmentation can result in, for example, overlapping jurisdictions, incompatible objectives, and unbalanced power relations (Visseren-Hamakers, 2015). Fragmentation of risk communication mechanisms can also undermine prevention and control efforts and public confidence (Jonsson *et al.,* 2016).

(6) Externalities

The negative impacts of invasive alien species often occur outside of the social or economic contexts responsible for their introduction and spread (**section 6.3.1.2**, also called telecoupling). For example, the cost of invasive alien species impacts are not included in the price of traded goods (Stoett, 2010). Invasion risk is sometimes not considered in the development of new agricultural and forestry technologies (Driscoll *et al.,* 2014), when deploying disaster relief aid or when developing international assistance programmes (Murphy & Cheesman, 2006); insect pests can be unintentionally imported with products used to rebuild infrastructure after natural disasters (**Chapter 3**, **section 3.2.2.2**). Negative environmental consequences of invasive alien species are often spatially and temporally diffuse, and this can undermine the legitimacy of environmental concerns (Neale & Macdonald, 2019). Biological invasion as an unintended consequence of trade is an example of a spill-over system, and spill-over effects can tend to be neglected in governance systems (J. Liu, Dou, *et al.,* 2018). The costs, liability and responsibilities for biological invasions need to be balanced between those directly responsible for species introductions and the general public, because health and biodiversity are a public good (i.e., nature's contributions to people and good quality of life; Outhwaite, 2010). One consequence of treating invasive alien species as an externality (**Glossary**) is that the welfare of the supply-side of trade is considered in isolation (D. C. Cook *et al.,* 2010). The trade-offs that occur as a result of unaccounted-for externalities result in conflicting interests (Hewitt & Campbell, 2007; Marire, 2015; Rouillard *et al.,* 2018; A. L. Smith *et al.,* 2013). Trade-offs also become increasingly political and difficult to resolve as they shift from within particular governance systems or sectors to between and outside of them (Visseren-Hamakers, 2015).

(7) Hurdles to implementation of policy

Although the arguments for invasive alien species policy implementation are empirically well supported, the extent and success of existing policy at international, national and sub-national scales is highly variable (**Figures 6.2**, **6.3**, **6.4** and **6.5**) and considered inadequate (Leadley *et al.,* 2014). Differences exist in the extent to which regulatory measures are implemented across countries (Brenton-Rule *et al.,* 2016) and laws are not always supported by regulation or implementation plans (Riley, 2012). There is no international authority, global coordination, or oversight mechanism for all invasive alien species, and the implementation of biosecurity practices under trade agreements is inconsistent (Stoett, 2010). There are several reasons why both policy and management implementation are challenging, including for example austerity measures and resource shortages (VanNijnatten, 2016), lack of capacity and expertise (Angulo & Gilna, 2008), as well as a number of the other challenges outlined in this section. Often there is a lack of monitoring to gather scientific evidence to support effective implementation, including information to evaluate the success of management approaches, such as knowledge generated from adaptive management (**Glossary**; Reed *et al.,* 2016; Smolarz *et al.,* 2016).

(8) The need for collective action

Effective prevention and control of invasive alien species can be achieved by cooperating and building of trust and social norms across actors, institutions, levels and sectors (**Figures 6.7** and **6.8**; McKiernan, 2018). However, conflicting interests and diverse values and perspectives mean that prevention and control programmes often fail (Graham *et al.,* 2019; Guerrero *et al.,* 2015; Smolarz *et al.,* 2016; **Chapter 1**, **section 1.5.2**; **Chapter 5**, **section 5.6.1.2**). For example, ineffective prevention and control can occur when individual land managers have an incentive to avoid invasive alien species control costs, thereby resulting in risks to others (Graham *et al.,* 2019). Similarly, actors that benefit from activities that increase risk of invasive alien species often have little incentive to acknowledge the risks and impacts of invasive alien species on other actors (Angulo & Gilna,

Primary driving forces **MANAGEMENT**	• Transport companies; Transport network designers; Organized individual businesses; Individual businesses • Trade industry; Tour operators; Agro-chemical sector; Consumer association; Local and regional authorities • Farmers and fishermen; protected area managers; forestry and horticulture managers; trade unions
Secondary driving forces **POLICY**	• EU institutions for Trade, Development, Environment, Climate, Water; Regional and National Authority • New countries; Sectoral associations for agriculture, horticulture, forestry, chemical industry, landscape planning • Social lobbies for consumers and business; Non-governmental organizations (NGOs); Media; Researchers
Tertiary driving forces **LIFESTYLE**	• Education systems; Research funders; Social lobbies and NGOs; Media and marketing industry • Politicians; Policy-makers; Consumers/Citizens • International/Multilateral institutions; Third countries
Base Short term **EXOGENOUS**	• General public and Migrant communities • National governments (demographic policy); European and national institutions and policymakers for climate • International/Multinational institutions; Third countries

Figure 6 8 **An example of the many relevant stakeholders for managing biological invasions from the Ebro River, Spain.**

There are four categories of stakeholders (on the left) and examples of stakeholders (on the right) involved in decision-making that affects the outcome of biological invasion, based on freshwater invasive alien species in the Ebro River, Spain. Adapted from Rodríguez-Labajos *et al.* (2009), https://doi.org/10.1016/j.ecolecon.2009.08.022, under copyright 2009 Elsevier B.V.

2008; Lubell *et al.,* 2017). Tensions between free trade and the governance of biosecurity risk further undermine the collective action (**Glossary**) that is needed on invasive alien species (Lansink *et al.,* 2018). The context-dependence of social settings and, therefore, of appropriate design of collaborative solutions, exacerbates this challenge (Lubeck *et al.,* 2019; P. Martin *et al.,* 2016). The transaction costs of administration, supervision and capacity development for implementing management of biological invasions can also undermine collective action (P. Martin *et al.,* 2016).

(9) Awareness, perception and values

There is often a lack of awareness and understanding, or neglect, of invasive alien species and their negative environmental, social and economic impacts. This is the case amongst a number of sectors, actors and stakeholders including amongst policymakers (**Chapter 1**, **section 1.5.2**; Moon *et al.,* 2015; Stoett, 2007). Perceptions of invasive alien species can also vary widely for several reasons (Shackleton, Richardson, *et al.,* 2019; Zengeya & Wilson, 2020), such as a focus on economic (instrumental) *versus* intrinsic and cultural (relational) values (**Chapter 1**, **section 1.5.2**; Leventon *et al.,* 2021). Understanding and the concept of biological invasion risk differs across and within groups and communities (Maclean *et al.,* 2022). Local communities, for example, may be very familiar with individual invasive alien species and their negative impacts on good quality of life, but remain unaware of the concept of biological invasions and the context of management and policy (Shrestha *et al.,* 2019; **section 1.6.7.1 (ii)**). Lack of awareness and different values can result in a lack of public support (Vane & Runhaar, 2016), exacerbate differing perceptions, and ultimately undermine collective efforts to manage biological invasions (Kohl *et al.,* 2019; **Chapter 3**, **section 3.2.1**). Public and private sector consultation and engagement can successfully underpin invasive alien species prevention and control efforts when they include effective risk communication and rely on the contributions of stakeholders and Indigenous Peoples and local communities (Ekanayake *et al.,* 2020; Falkenmark, 2007; Jonsson *et al.,* 2016).

(10) Conflicting interests and trade-offs

Conflicting interests and trade-offs happen when externalities are not considered and when a priori risk assessment is not done (e.g., Driscoll *et al.,* 2014), or is not inclusive of sectors and stakeholders (Woodford *et al.,* 2016; Zengeya *et al.,* 2017). Since some invasive alien species can have both positive and negative impacts, control and eradication programmes can spark conflict (**Chapter 1**, **Figure 1.1**, **section 1.5.2**; **Chapter 4**, **section 4.1.2**). There is also the often challenging need to strike a balance between short term needs and the long-term maintenance of good quality of life, alongside the uncertainty and challenges in making predictions outlined above.

6.2.3 Options for strengthening the governance for biological invasions

"Current environmental challenges call for new interdisciplinary approaches at the interface of natural and social sciences, framed in a context of governance and decision-making by actors from the state, market and civil society" (Padt *et al.,* 2014).

The challenges outlined in the previous section provide a platform for identifying a range of options for strengthening the governance for biological invasions. Governance-related options that emerged from the literature review[5] are summarized in **Table 6.6** under six general topics: strategy; multilevel and multisector governance; coordination and cooperation; policy environments that are enabling; research and information and their communication; and governance capability, capacity and resourcing. Governance provides an overarching instrument for dealing with complex systems and is considered one of the most important factors to achieve desired environmental outcomes (Bennett & Satterfield, 2018). The many considerations and elements of

5. Data management report available at: https://zenodo.org/doi/10.5281/zenodo.5762739

Box 6 4 **Restoring the Kafue flats: a case study of integrated management and effective governance of the invasive shrub, *Mimosa pigra* in Zambia.**

Impact on good quality of life and protected areas: In the early 1980s, *Mimosa pigra* (giant sensitive plant) invaded the Kafue Flats – a 6,500 km^2 floodplain located in the southern region of Zambia and a designated Wetland of International Importance under the Ramsar Convention. The flood plain is world-renowned for its abundant floodplain wildlife, including the endemic antelope, *Kobus leche kafuensis* (Kafue Lechwe), and rich diversity of birdlife including the *Bugeranus carunculatus* (wattled crane) and *Balearica regulorum* (gray crowned-crane). The flood plain hosts two national parks, surrounded by buffer zones inhabited by smallholder farmers and fishers, whose livelihoods depend on the land and water resources of the floodplain. It has been suggested that the *Mimosa pigra* invasion was triggered by hydrological alterations resulting from hydropower dams (Blaser, 2013; Mumba & Thompson, 2005). What started as a small infestation of about

Box 6 4

2 ha spread rapidly and covered over 3000 ha of the floodplain (Blaser, 2013; Shanungu, 2009; Solomon Genet, 2007; Thomas, 2007). Consequently, the native floodplain vegetation was replaced by *Mimosa pigra*, and many wildlife species of conservation concern including the *Kobus leche kafuensis* and *Bugeranus carunculatus* were displaced as their habitat shrank (**Glossary**; Blaser, 2013).

Multiparty governance: In 2017, The International Crane Foundation/Endangered Wildlife Trust Partnership (ICF/EWT Partnership), World Wide Fund for Nature (WWF) – Zambia, and Zambian Department of National Parks and Wildlife (DNPW) embarked on a three-year cooperative project to address the continued spread of *Mimosa pigra*, restore the floodplain grasslands and enhance their ability to support important biodiversity of the flats, and control up to 95 per cent of the baseline cover of *Mimosa pigra* in an effort with substantial community involvement.

Stakeholder and Indigenous Peoples and local communities involvement and benefits: The project also focused on developing the Zambian Department of National Parks and Wildlife local staff capacity in invasive plant management and habitat restoration. This included ecological research to enhance global understanding of large-scale *Mimosa pigra* control methods and their measurable impact on biodiversity, livelihoods options, and the broader economy through agriculture, fisheries and tourism **(Figure 6.9)**. The project took an ecosystem approach with a focus on the wider Kafue Flats ecosystem – including the two National Parks and the buffer zones – with a strong emphasis on a multi-sector approach in the management of the invasive alien species.

The project intended to engage non-traditional stakeholders including the private sector. The integrated management approach adopted combined physical, chemical and biological control options (**Glossary**).

Sustainable successes: By 2020, all management options described above had been implemented. About 450 workers from local communities were employed to undertake community-based restoration work through large-scale physical removal and chemical spraying of *Mimosa pigra*. Biological control trials through the importation and direct release of the control agent *Carmenta mimosa* commenced in May 2019. Six months after direct release, a monitoring exercise was undertaken to determine if there were signs of the control agent's survival. The presence of adults and actively feeding larvae six months after the introduction indicated that the biocontrol agent survived successfully and was reproducing. By June 2020, the area invaded by *Mimosa pigra* had been reduced by approximately 68.8 per cent of the total invaded area at baseline. Ecological surveys indicate that there is regeneration of native vegetation in areas previously covered by *Mimosa pigra* and use of the restored sites by herbivores including the *Kobus leche kafuensis*, as well as various species of resident and migratory waterbirds including breeding pairs of *Bugeranus carunculatus* and *Balearica regulorum*. Some members of the community employed by the project had been able to use their income to invest in livestock and improved housing while others had used it to educate their children.

Long-term efforts were undertaken to restore water conditions through environmental flow releases from the dam upstream that might limit future *Mimosa pigra* establishment.

Figure 6 9 ***Mimosa pigra*** **(giant sensitive plant) in Zambia.**

Left: Clearing of *Mimosa pigra* from the Kafue flats floodplain. Right: control of *Mimosa pigra* maintains habitat for the endemic *Kobus leche kafuensis* (Kafue Lechwe) and other biodiversity in this wetland of international importance. Photo credits: Gareth Bentley (WWF Zambia) – Copyright (left) / Patrick Bentley (WWF Zambia) – Copyright (right).

successful governance for biological invasions are well illustrated by the case of the invasive alien shrub, *Mimosa pigra* (giant sensitive plant), in Zambia **(Box 6.4)**.

As context for the sections to follow, three key points can be made about the literature and evidence in support of governance approaches for biological invasions:

- While there is literature on the topic of biological invasions governance (assessed and drawn upon in the formulation in this section), only a small proportion of this literature critically evaluates, with empirical data in an invasive alien species context, the strengths and weaknesses of particular or alternative governance models and overarching governance systems.
- The most useful and evidence-based literature comes from comparatively extensive and relevant work on environmental governance more broadly.
- As a result, many of the options, tools and approaches are not particular to biological invasions and evidence for them is steeped in different domains and areas of expertise (Weitz *et al.,* 2017). Since an assessment of environmental governance is beyond the scope of this assessment – the field is interdisciplinary and itself developing rapidly – this section draws on some of the general frameworks and thinking on environmental governance that align with insights from biological invasion-specific literature, and refers to key findings in depth only where there is evidence specific to biological invasions.

Table 6 6 **Overview of governance response considerations.**

This table presents six types of response options (left column), and examples of relevant support tools, methods and frameworks (middle column) alongside examples of publications (right column).

Response option	Examples of relevant support tools, methods and frameworks	Example publications
Strategy, including approaches to deal with inherent complexity **(sections 6.2 to 6.7)**	• Empirical analysis of invasive alien species policy and governance • Objective review and evaluation • National invasive alien species Strategy and Action Plans • Sustainability: Environmental – social – economic • Pressure-State-Response type models • Ecosystem-based approach to management (EBM) • Adaptive governance (**Glossary**) model	Barnhill-Dilling *et al.,* 2019; Boström *et al.,* 2016; Chaffin *et al.,* 2016; Cooney & Lang, 2007; P. Martin *et al.,* 2016; McGeoch, Shaw, *et al.,* 2015; Rudd *et al.,* 2018; Smolarz *et al.,* 2016; Termeer *et al.,* 2010
Multi-level and sectoral integration **(sections 6.3, 6.7)**	• Integrated governance for biological invasions • Transnational environmental alliances • Conflict resolution • Negotiation of values • Inter-agency coordination to co-ordinate across policies and agencies and to monitor (stakeholder and Indigenous Peoples and local communities-wide or) government-wide activity. • Multidisciplinary, comparative research on invasive alien species policy regimes (**Glossary**)	Bennett & Satterfield, 2018; Daviter, 2017; Herrick, 2019; Justo-Hanani & Dayan, 2020; Visseren-Hamakers, 2015; Weitz *et al.,* 2017
Coordination and collaboration across international and regional mechanisms **(section 6.2.3.4)**	• Stakeholder and Indigenous Peoples and local communities mapping • Actor network analysis • Measures to build public support • Bridging organizations (**Glossary**) • Extension personnel • Institutions that build cooperation amongst relevant actors • International cooperation on information sharing, monitoring, implementation and best practice	Angulo & Gilna, 2008; D. C. Cook *et al.,* 2010; Gilna *et al.,* 2014; Lubell *et al.,* 2017a; Nourani *et al.,* 2019; Stoett, 2010
Policy that is enabling, including the consideration of inclusion, the distribution of power and adaptation **(sections 6.4, 6.5)**	• Policy risk analysis • Assess the distribution of costs and benefits of governance actions • Stakeholder and Indigenous Peoples and local communities mapping • Ecosystem Based approach to Management (EBM) • Mechanisms to identify the need for and enable the establishment of temporary task forces • Networked, polycentric governance (**Glossary**) • Legitimize decision-making at local scales • “Landcare” model	Catacutan *et al.,* 2009; Chaffin *et al.,* 2016; Linke *et al.,* 2016; Marshall *et al.,* 2016; P. Martin *et al.,* 2016; P. Martin & Taylor, 2018; McKiernan, 2018; Moon *et al.,* 2015; Peltzer *et al.,* 2019; Smolarz *et al.,* 2016

Table 6 6

Response option	Examples of relevant support tools, methods and frameworks	Example publications
Effective communication of research, information and learning **(section 6.6)**	• Biosecurity collectives for information sharing • Structured process by which knowledge can influence relevant actors • Knowledge sharing platforms and infrastructures at multiple scales • Clear assignment of responsibilities for risk communication • Public information campaigns • Context-specific messaging to encourage strategic behaviour • Information brokers	D. C. Cook *et al.,* 2014; Cooney & Lang, 2007; Jonsson *et al.,* 2016; Lubeck *et al.,* 2019, 2019; Moon *et al.,* 2015; Nourani *et al.,* 2019
Governance capability, resourcing and capacity **(sections 6.2 to 6.6)**	• Build capacity in key governance capabilities • Campaigns to make necessary technical concepts part of the public agenda • Consider gains and losses from activities (e.g., trade) in negotiations • Cost sharing arrangements • Assess potential inequity and incapacity	D. C. Cook *et al.,* 2014; Ford-Thompson *et al.,* 2012; Jonsson *et al.,* 2016; P. Martin *et al.,* 2016; P. Martin & Taylor, 2018; Outhwaite, 2017; Termeer *et al.,* 2016; Termeer & Dewulf, 2014

6.2.3.1 Employing multiple models of governance

Together, four complementary models of governance (1-4 below) provide a high-level framing for comprehensive governance and for guiding the development of national invasive alien species strategies (**Figure 6.6**; IPBES, 2019b). These models provide alternative but, importantly, not mutually exclusive mechanisms for bringing about policy implementation, and together they encompass a focus on all relevant actors (Primmer *et al.,* 2015; **Figure 6.7**). Each of these models thus plays a role in the comprehensive and strategic governance for biological invasions; each encompasses options for strengthening governance that are outlined in further detail in **sections 6.3** to **6.6**.

(1) Hierarchical governance: Top-down governance based on agreed invasive alien species -relevant policies or decisions and their implementation

Governments enact legislation, develop aligned regulatory policy, and provide the funding needed to implement risk assessment and surveillance (**Glossary**; Lodge *et al.,* 2006), i.e., provide a comprehensive, centralized and science-based control regime, administered through one or more national agencies (Herrick, 2019). Hierarchical governance provides an existing and necessary backbone as well as, *via* legislation and regulation, the strongest category of instruments for invasive alien species implementation and control. While shortcomings in the hierarchical governance for biological invasions are identified and discussed above and in multiple sections of this assessment (**section 6.2.2.(4)**), hierarchical multilateral and national policy and legislative instruments will remain central to governance for biological invasions (**section 6.3**).

(2) Strategic-behavioural governance: Governance encouraging actors to behave in the collective interest of limiting invasive alien species and their harm

Beyond legislated policy, broad stakeholder and Indigenous Peoples and local communities support is essential to the effectiveness of invasive alien species prevention and control, including the full breadth of relevant actors (**Figures 6.7** and **6.8**; Vane & Runhaar, 2016). Strategic institutional arrangements can create enabling environments for collaboration, achieving agreement, and enhancing effective action. Perceived costs and risks of invasive alien species as well as opposition based on moral or ethical considerations can undermine management outcomes, whereas community groups, lobbies and public support can be particularly powerful in altering actions that affect invasive alien species outcomes (Crowley *et al.,* 2019; P. Martin *et al.,* 2016). Public support and voluntary, collective action are needed, for example, to manage weeds that cross boundaries. The willingness of land owners to participate in interventions is determined by many individual, collective and context-specific factors (Finkel & Muller, 1998; Lubeck *et al.,* 2019; Vane & Runhaar, 2016). The research and design of tailored behaviour-change strategies, effective communication and outreach and the analysis of policy risk to anticipate undesirable outcomes are all key components of successful strategic-behavioural governance (Lubeck *et al.,* 2019). The net balance of incentives and disincentives determines the likelihood of participation in invasive alien species prevention and control efforts. The focus of strategic-behavioural governance is therefore on social and economic mechanisms for bringing about public support and behavioural change (Martin *et al.,* 2016; **sections 6.4** and **6.5**).

(3) Scientific-technical governance: Governance of knowledge to systematically support decision-making

Effective governance for the prevention and control of invasive alien species demands a wealth of information, efficient delivery of this information, and context-appropriate means by which to communicate it. Scientific-technical governance deals with the governance of knowledge within and across the components of the socioecological system that characterizes biological invasion (**Figure 6.7**; McGeoch & Jetz, 2019). This includes the role of international collaboration in delivering and sharing knowledge (Latombe *et al.,* 2017) and regional early warning and information systems for invasive alien species. Scientific-technical governance includes the structure of information systems and platforms, assignment of responsibilities for data and information generation, sharing and communication (including risk communication; **Chapter 5**, **section 5.2.2.1.h**). It also includes strategies for delivering and communicating different types of information to different stakeholders. For example, the European Commission has developed an invasive alien species information system (European Alien Species Information Network, EASIN) that ensures transparent and authoritative data on invasive alien species (European Environment Agency, 2010a). Scientific-technical governance could also involve introducing, or strengthening existing, mechanisms that support a more ecosystem-based approach to governance, i.e., that includes systematic collection of essential data, use of best available evidence, and impact assessments as a pre-condition for new activities, policy change and involvement of stakeholders (Smolarz *et al.,* 2016).

(4) Adaptive-collaborative governance: Communication across sectors and governance levels to enhance shared invasive alien species goals

This model of governance involves a systematic approach to improve the planning and management of invasive alien species by "learning from doing". It involves joint formulation of management objectives, specification of multiple management options, forecasting and estimating uncertainty, implementing management options, monitoring (social learning) to improve forecasting and reduce uncertainty, and changing management responses throughout a policy cycle (**Glossary**; Niemiec *et al.,* 2019; Richardson *et al.,* 2020). To date, many approaches to governance for biological invasions that refer to adaptive management have included only scientists, other experts or formal invasive alien species managers, and top-down modes of governance. In contrast, the concept and practice of adaptive collaborative governance and management are based on the involvement of stakeholders in decision-making at all levels, and on the establishment of vertical and horizontal institutional linkages spanning governance scales. These linkages support integrating and sharing knowledge. Adaptive-collaborative governance and management is ultimately "concerned with enhancing and including the capacity of all actors with a stake for sustainably managing the resource at hand" (Plummer *et al.,* 2012). Options involving this model of governance are covered in further detail in **section 6.4**.

6.2.3.2 Developing effective strategy for biological invasions

Recognizing the significance of strategic planning for invasive alien species, one indicator under the SDGs (Indicator 15.8.1) aims to track the percentage of countries with national strategies for preventing and controlling invasive alien species (UN, 2021). The need for strategy to deal with biological invasions is driven by:

1. The sheer size of the problem and the need to prioritize resources and actions;

2. The multidimensional and interconnected nature of the problem across invasion stages (**Glossary**), sectors and actors; and

3. The interdependence between invasion and other forms of environmental change.

Strategic planning

The way strategies are designed, their content and the incorporation of good and environmental governance principles are key to guarantee their effectiveness (**Chapter 5**, **section 5.2** for more information on evidence-based decisions). In complex contexts, such as those faced by countries dealing with invasive alien species and their impacts, strategic planning can be improved by clear and cyclical assessment, option formulation, action, and re-assessment to achieve the goals of prevention, control and minimization of negative impacts (Andonova & Mitchell, 2010), including regular, objective review and evaluation (Martin *et al.,* 2016; **Table 6.6**). Given limited resources, strategic planning can drive prioritization, including determining which species need prevention, control, or adaptation responses (McGeoch *et al.,* 2016). The strategic planning phase would consider all four governance models discussed above as part of a comprehensive strategy. Widely accepted steps in the development of strategy include: (1) evidence-based situation analysis; (2) development of a strategy and action plan; (3) identification and prioritization of tools and methods to enable strategic action, including legislation, financing, institutional arrangements, stakeholder and Indigenous Peoples and local communities participation; and (4) mechanisms to ensure implementation (Falkenmark, 2007).

National strategies

National strategies are critical for achieving invasive alien species goals and targets, as this is the level at which legislative and resourcing commitment by countries is strongest (CBD, 2020c). National strategies for invasive alien species have been called for, *inter alia*, to design implementation regimes, for example in the form of national invasive alien species strategy and action plans or national-level biosecurity strategies (Sustainable Development Solution Network, 2021). Such strategy could aim to include or address:

- The means to achieve coherent legislative frameworks;
- Coordination mechanisms to manage and communicate with the range of government and non-government sectors and actors involved;
- A coordinating body able to harmonize law such that no conflicts exist between sectors (Riley, 2012; Shine *et al.,* 2005);
- Collaborative and inclusive definition of goals and objectives for invasive alien species across sectors and levels that can be integrated into national strategies (Barnhill-Dilling *et al.,* 2019; Praseeda Sanu & Newport, 2010; Smolarz *et al.,* 2016);
- The identification, prioritization and management of pathways and drivers;
- Prioritizing established and future invasive alien species threats and committing related resources accordingly;
- Optimizing surveillance, early detection and rapid response, eradication, containment and control programmes at local and sub-national scales;
- The prioritization of national strategies to improve the efficiency of deployment of limited resources for invasive alien species prevention and control;
- National strategies can also define instruments and processes to encourage shared efforts and commitments, and understanding of the specific roles of all sectors and actors (Indigenous Peoples, community and industries) and multi-scale coordination of response programmes (e.g., Maclean *et al.,* 2021);
- Mechanisms for specifying the distribution of responsibility (financial, planning, infrastructure, etc.) amongst stakeholders (Smolarz *et al.,* 2016);
- Coordination and justification for efficient and effective investment (whether national and sub-national) and appropriate support and reporting on invasive alien species guiding principles **(Table 6.3)**, guidelines, goals and targets under multilateral agreements, in the context of societal and economic goals of sustainable development and international trade;
- Mechanisms to drive institutional and organizational structures that allow for flexible strategic thinking and reflection (Boström *et al.,* 2016) and adaptive cooperation between stakeholders (Smolarz *et al.,* 2016);
- National strategies that address the need for and design of, local and subnational strategies for the eradication of priority species.

6.2.3.3 Including actors across scales, levels and sectors

There is increasing evidence and a growing realization that the involvement of multiple sectors, stakeholders and Indigenous Peoples and local communities, together with the consideration of diverse perspectives and interests, can achieve effective governance and management of biological invasions (Guerrero *et al.,* 2015); thus, a sustainability framework for an invasive alien species strategy could be appropriate (e.g., Barnhill-Dilling *et al.,* 2019; Vaas *et al.,* 2017). Governing invasive alien species within a sustainability framework provides a widely-accepted departure point for national environmental strategies (Nourani *et al.,* 2018), including strategies for invasive alien species. Including stakeholders and Indigenous Peoples and local communities with different knowledge, perceptions and socio-cultural contexts can help achieve shared efforts and commitments, the understanding of the specific roles of all actors, improve the efficiency of proposed mechanisms and build trust (Maclean *et al.,* 2022; Shackleton, Richardson, *et al.,* 2019). In other words, everyone has a role to play in the governance for biological invasions.

Such joint or integrated approaches (**section 6.2.4**) across the components and processes that characterize the socioecological system relevant to biological invasion (i.e., multilevel and multisector governance) can improve the effectiveness and efficiency with which the complexity of the invasive alien species problem can be managed (Lubeck *et al.,* 2019; Stoett, 2007; **Figure 6.7**). While the terms "level" and "scale" are often used interchangeably, they have distinct, complementary meanings in governance for biological invasions. Because of their importance to the design of effective governance systems, these dimensions and the roles that they play are outlined below and discussed in terms of their importance for sectors and networks.

Scales and governance – spatially and temporally continuous structures and processes

One way of viewing governance is through a scaling lens (Padt & Arts, 2014). While the impacts of invasive alien species occur locally, the drivers that facilitate biological invasions operate across scales from global to local, and the impacts also accumulate upwards to affect national and global economies and ecosystem processes (Andonova & Mitchell, 2010; Boström *et al.,* 2016; Termeer & Dewulf, 2014). As a global change phenomenon, biological invasion is both complex and dynamic because it involves interacting social, biological and abiotic environmental dimensions, often with context-specific outcomes (**Chapter 1**, **section 1.5**). Biological invasions are also transboundary in nature. This is a consequence of the fact that species movements are not naturally constrained by geopolitical boundaries: borders can be fluid for stakeholders and Indigenous Peoples and local communities and trade and human movement across natural and geopolitical borders are the primary drivers promoting biological invasion (S. Muller *et al.,* 2009). As a result, solutions for managing biological invasions demand strategy, communication, cooperation, data and information that are similarly geopolitically unbounded **(Figure 6.7)**. Agencies responsible for management are often local and the transfer of knowledge and management technology to this level is crucial (**section 6.6**). Biological invasions and management events at one place or time, and the reporting of such events, have a fundamental bearing on relevant response options at scales beyond which they occur. The process of biological invasion operates continuously across spatial scales from local – sites at which populations of invasive alien species establish or have impact – to the large regions over which invasive alien species are transported, cross borders and spread. Similarly, invasive alien species management spans short-term actions – such as rapid responses to eradicate newly established invasive alien species – to long-term efforts to contain or control well-established invasive alien species in order to mitigate their impacts. Investing in invasive alien species management systems is a long-term endeavour to protect and maintain good quality of life and nature's contributions to people. Therefore, it is appropriate, for example, that invasive alien species information systems, management of invasive alien species, and governance for biological invasions structures account for such scales. All of these information, structures and processes are planned and implemented across multiple spatial and temporal scales of biological organization (i.e., considering genetic diversity and adaptation, species population dynamics, community processes and ecosystem function; Padt & Arts, 2014).

Levels of governance – vertical interactions

Invasive alien species are governed and managed at multiple levels of societal organization, from regional to national and sub-national (**Chapter 1**, **Figure 1.9**). Levels of governance encompass civil society groups, for example, that contribute to weed clearing in local neighbourhoods, to sectoral land-management at a sub-national scale (such as protected and production areas), to states and provinces, countries, regions and broader intergovernmental arrangements (**Figures 6.7** and **6.8**). When a mismatch exists between the level of governance and the scales at which biological invasion occurs, policy and resulting interventions are less likely to succeed (Primmer *et al.,* 2015). Biological invasions policy is relevant and necessary at all levels of governance, and specification of those levels is useful, if not essential, in strategic planning and decision-making (Lescrauwaet *et al.,* 2015). Stakeholders and institutions affected by and responsible for governance for biological invasions operate across either more or less hierarchical or inclusive levels of responsibility and cooperation. At a sub-national level, there are several possible invasive alien species management institutions, such as state/province-wide management programmes (bounded by sub-national government borders), cooperative management areas (delineated by land use or ownership) and volunteer groups (**Figures 6.7** and **6.8**). For example, the Landcare movement across multiple IPBES regions provides a tested option for government-supported, community-led information sharing and action, including partnerships among business, researchers, natural resource management agencies, governments, stakeholders and Indigenous Peoples and local communities, resulting in several successful cases of local implementation (Catacutan *et al.,* 2009; McKiernan, 2018).

Sector governance – horizontal interactions

Invasive alien species prevention and control activities, including legal and regulatory instruments for biological invasions, involve multiple institutions with global (CBD, WTO, IMO) or regional (Council of Europe) mandates and tend to be developed for and organized within key industry sectors (Hulme, 2020). As discussed further in **section 6.3.1**, these sectors include environment and biodiversity, transport, trade, production systems, extraction systems and public health. One main limitation of the current policy regime for managing biological invasions is the narrow sectorial focus, where legal and regulatory instruments focus only on addressing either biosafety or biodiversity issues. The need for information flows and communication across governance systems from different sectors has been identified as a major challenge that undermines the effectiveness of invasive alien species management (Roura-Pascual *et al.,* 2021; **Chapter 5**, **section 5.6.2.2**) and a limitation for effective horizontal integration of invasive alien species management approaches. Moreover, many of these sectors influence public policy and resources (notably production, extraction, development aid and health sectors), so the explicit consideration and inclusion of all sectors is critical for effective governance for biological invasions.

Network governance – horizontal and vertical interactions

From an analytical perspective, and with the purpose of better understanding the roles and interactions among actors, scales, levels and sectors **(Figure 6.7)**, governance systems can be considered as networks (Lubell *et al.,* 2017; Provan & Kenis, 2008; **Chapter 5**, **section 5.6.3.1**). Networks are a useful way to jointly consider the scales, levels and sectors outlined above. For example, the multiple relationship links between stakeholders needed to manage aquatic invasions in rivers in **Figure 6.8** can be viewed and as a network to better understand the strengths, weaknesses and gaps in governance for biological invasions in this freshwater context. A network approach is useful for understanding the roles and contributions of stakeholders and institutions for cooperation and for strengthening the effectiveness of working relationships (Moon *et al.,* 2015; VanNijnatten, 2016). A network view of governance for biological invasions encompasses the concept of polycentric governance (one with multiple centres of power in decision-making) that has been identified as a successful and complementary model for inclusive governance for biological invasions (Marshall *et al.,* 2016; Vaas *et al.,* 2017). Some of the advantages of polycentric governance include better information generation and flow within and across actors (nodes) in the network compared to monocentric governance, as well as short social and physical distances between interacting nodes (Cook *et al.,* 2010, 2014; Vaas *et al.,* 2017; **section 6.4.4**).

6.2.3.4 Coordination and cooperation to support the governance for biological invasions

Regardless of the view taken (scaled, multilevel or multisector, networked, or integrated), governance for biological invasions is achieved through cooperation, coordination and effective communication (Jacobs, 2017; Lubell *et al.,* 2017; McKiernan, 2018; Vaas *et al.,* 2017). Options for enabling integrated governance for biological invasions thus include identifying and supporting stakeholders who are able to play a bridging role across otherwise disconnected nodes of the network, the establishment of formal coordination bodies, and the use of extension personnel (Ekanayake *et al.,* 2020; Nourani *et al.,* 2019; Vaas *et al.,* 2017; **sections 6.2.4** and **6.4**). International networks and partnerships play a decisive role in sharing information, capacity-building, promoting collaboration, and sharing novel tools and techniques to manage biological invasions (**Chapter 5**, **section 5.6.3.1**). For example, sharing of knowledge between native and invaded ranges (**Glossary**) helps to predict entry and establishment risks and the potential impacts of alien species (Nourani *et al.,* 2018). International collaboration is critical in managing biological invasions since the alien species are mobile and do not respect political or legal boundaries (Graham *et al.,* 2019).

Governance for biological invasions is therefore in part a collective action problem that provides collaborative solutions (Hershdorfer *et al.,* 2007; Epanchin-Niell *et al.,* 2010; McLeod & Saunders, 2011; Bagavathiannan *et al.,* 2019) including, for example, public-private partnerships (Mato-Amboage *et al.,* 2019). As outlined earlier, the mobility of invasive alien species means that preventing spread and managing established populations can be achieved through cooperation and coordination across property and jurisdictional boundaries (Graham, 2014; Yung *et al.,* 2015; Howard *et al.,* 2018; **section 6.4**). Achieving such cooperation is challenging because diverse actors have different perceptions and values (**Chapter 1**, **section 1.5.2** and varying levels of interest, skills, resources, capacity, and time to commit to invasive alien species prevention and control (Donaldson & Mudd, 2010; Graham, 2013; Ma *et al.,* 2018; Kropf *et al.,* 2020). Successful collective action would include developing stakeholder and Indigenous Peoples and local communities networks, and building the trust to forge a common understanding of the problem, agree on a common goal, identify measures of success, and encourage participation in individual and group activities (Stallman & James, 2015; Niemiec *et al.,* 2016; Graham & Rogers, 2017; T. M. Howard *et al.,* 2018; Bagavathiannan *et al.,* 2019). Micro-interventions implemented during community engagement activities can increase participation and change social perceptions, such as facilitating increased communication amongst community members, setting collective goals, achieving public commitment, and enhanced visibility of contributions (Niemiec *et al.,* 2019). There are many examples of how local communities have successfully mobilized to collectively manage invasive alien species (**section 6.4.3**).

6.2.3.5 Considering human adaptation to invasive alien species in governance systems

Adaptation to invasive alien species is emerging as a critical consideration for policy and management. Two concepts of adaptation are relevant: the first is "planned" adaptation, derived from a concept used by the Intergovernmental Panel on Climate Change (IPCC), which is "the result of a deliberate policy decision, based on an awareness that conditions have changed or are about to change and that action is required to return to, maintain, or achieve a desired state" (IPCC, 2007); the second is "autochthonous" adaptation, defined as "deliberate adaptation actions undertaken by individuals or small social groups that are specific to and occur within a local system, where human populations are ultimately affected" (P. L. Howard, 2019). This type of adaptation has four characteristics: (1) it is deliberate; (2) it refers to individuals and small groups of individuals; (3) it is specific to the locality – specific environmental, social and cultural conditions that prevail in specific places where people live and act and (4) it occurs within a local system, which is affected by multi-scalar drivers and feedbacks, thus it is affected by many external influences, including planned adaptation (P. L. Howard & Pecl, 2019).

When human adaptation becomes a response

Adaptation is relevant in cases where invasive alien species are established and, for environmental, management, or socio-economic reasons, there may currently be no other option (Kleinschroth *et al.,* 2021). It is also relevant when invasive species impact human well-being and people attempt to manage them or adapt to their impacts (König *et al.,* 2020). Adaptation may be the only option in cases where, due to the type of invasive and invasion scale, resources are unavailable to effectively mitigate or control invasive alien species, such as in forests, rangelands, savannahs, and large water bodies, including oceans (Godfree *et al.,* 2017). It may also be necessary in cases where there are currently no known effective control methods, or effective methods cannot currently be deployed due to non-target effects or strong political, ethical, or social objections to available controls methods. For example, recreational fishing lobbies can stand in the way of formulating invasive alien species regulations (Zengeya *et al.,* 2017; **Box 6.16**). Adaptation may be the only option when invasive alien species generate substantial social, economic or ecological benefit and have been incorporated into socioecological systems to such a degree that control or eradication would generate serious negative socioecological impacts (Bhattacharyya & Larson, 2014; P. L. Howard, 2019; Roder, 2001). When invasive alien species have negative impacts on good quality of life (**Chapter 4**, **section 4.5**), people attempt to change these impacts and, if possible, turn harm to benefit. A review of 70 case studies on adaptation to invasive alien species across the globe found that this is done in many ways – such as managing invasive alien species, using invasive alien species, changing their cropping and livelihood systems to accommodate harmful changes, or using the resources that invasive alien species can in some cases provide (P. L. Howard, 2019). When the impacts are too severe, people may be forced to abandon their homelands altogether or to migrate to find resources such as forage grass in new regions (**Chapter 4**, **sections 4.5.1** and **4.6.3.2**). Adapting to invasive species, then, often means mobilizing and reorganizing relationships and assets within communities, which has knock-on effects not only for individual members but as well for entire communities and socioecological systems (P. L. Howard, 2019).

Governance implications of human adaptation to invasive alien species

The practical and policy implications of such local-level adaptation to invasive alien species are significant. No matter how wide the reach of planned interventions, such adaptation may still be necessary. Governments often have limited resources and rely on local actors and their cooperation to implement invasive management actions (P. L. Howard & Pecl, 2019; Pecl *et al.,* 2019). Adaptations to invasive alien species occur in different spheres of individual, household, or collective activity related to production systems and the enactment of daily life (P. L. Howard, 2019). Local-scale adaptation is an important means to mitigate the impacts of invasive alien species, restore socioecological resilience (**Glossary**) and, where necessary and possible, transform socioecological systems to more desirable and sustainable states. In cases where adaptation includes use of the invasive alien species as a resource, a balance needs to be achieved between local benefits arising from such use and the potential of such use exacerbating negative outcomes from further invasive alien species spread (P. L. Howard & Pecl, 2019).

Understanding and considering human adaptation to invasive alien species can lead to the formulation of policies and practices (related to sectors such as land management and pesticide use) that seek to influence local adaptation in ways that increase adaptive capacity, resilience and sustainability. It is therefore an important, although to date little considered, phenomenon and can be considered as a viable response option in inclusive, integrated governance for biological invasions (**section 6.2.3.5**).

6.2.4 Integrated governance for biological invasions

Drawing on the approaches above and recognizing the relevance of multiple scales, sectors and levels of governance, "Integrated Environmental Governance" provides an option for improving the effectiveness of invasive alien species prevention and control because it focusses attention on the relationships between the necessary components of governance systems for biological invasions **(Box 6.5)**. In this way, context-specific application of integrated governance potentially simultaneously helps to address the challenges of fragmentation, complexity and information flow that are currently pervasive in governance

Box 6 5 Integrated governance for biological invasions.

Integrated governance for biological invasions consists of establishing the relationships between the roles of actors, institutions and instruments, and involving as appropriate all those elements of the socioecological system that characterize biological invasion and its management, for the purpose of identifying the strategic interventions needed to improve invasive alien species prevention and control outcomes.

for biological invasions (**section 6.2.2**). Coherent and better integrated policy regimes (**Glossary**) have been called for that aim to enable more effective and efficient policy outcomes, reduce policy conflicts, implementation delays, confusion and lack of clarity for stakeholders, wastage of resources and unanticipated outcomes in the complex contexts that characterize the governance and management of biological invasions (Riley, 2012; Vaas *et al.,* 2017). The definition of integrated governance for biological invasions below **(Box 6.5)** is in line with and built upon the concept of integrated environmental governance (Visseren-Hamakers, 2015; Visseren-Hamakers *et al.,* 2021)

Integrated governance, including for biological invasions, includes not only integration across sectors (so-called "nexus" in sustainable transitions literature; S. Díaz *et al.,* 2019; **Glossary**; **Chapter 1**, **Box 1.14**), but also a range of strategic actions and governance system properties characterize good governance for biological invasions (Weitz *et al.,* 2017). In other words, policy integration is only one part of the integration needed, and attention may also be given to the properties of the broader system that delivers invasive alien species policy (Leventon *et al.,* 2021).

A key part of the recognition of the need for integrated governance for biological invasions concerns the need for integration across the sectors that in some way intersect with the problem of invasive alien species – as either causal, affected or managing actors **(Figure 6.7)**, as discussed above. These sectors include environmental, human, animal and plant health (Hulme, 2020). This approach is referred to more broadly in governance literature as the "nexus approach" (S. Díaz *et al.,* 2019; Weitz *et al.,* 2017), and it "focuses on the relationships between different policies and sectors (e.g., agriculture, transport, environment) with the aim of coordinating across sectors without preferring one over the other in order to promote coherence" (Visseren-Hamakers, 2015). The intention of such integration is to improve policy coherence by "identifying synergies and trade-offs, optimizing policy options, and adapting governance arrangements" (Weitz *et al.,* 2017). This approach aims therefore to reduce undesirable outcomes for invasive alien species management that result from conflicting policy and interests across sectors. An example is the "One Biosecurity" approach: "an interdisciplinary approach to biosecurity policy and research that builds on the interconnections between human, animal, plant and environmental health to effectively prevent and mitigate the impacts of invasive alien species" (Hulme, 2020; **Glossary**).

While considering what needs to be integrated (e.g., research, sectors, policy) and how (e.g., stakeholder and Indigenous Peoples and local communities' inclusion, analysis, collective action) holds significant promise to achieve better outcomes for invasive alien species prevention and control, there are also limits to this approach that are important to recognize. The status of biological invasions and the most effective management approaches are to a large degree context-dependent, therefore, comprehensive and strongly centralized policy integration may be neither possible nor desirable (Herrick, 2019; Hoff *et al.,* 2019). New approaches or decision-making structures are not developed from a clean slate and can be strategically designed to strengthen or fill gaps in existing governance systems (Visseren-Hamakers, 2015).

Nonetheless, there is substantial evidence to suggest that a greater degree of policy integration would be beneficial in many instances (Lansink *et al.,* 2018; Smolarz *et al.,* 2016), and the benefits to building on existing policy settings have been highlighted (Trump *et al.,* 2018). To this end options for more effective invasive alien species policy, including integration where it is needed, include a number of desirable features, such as policy coherence and political legitimacy (Daviter, 2017; Herrick, 2019). Several tactics that enable this reform can be incorporated into invasive alien species strategies at national and other levels and sectors **(Table 6.7).** Other key considerations include external influence and dealing with the factors that influence integration beyond cross-sector relationships and policy (**section 6.2**). Finally, negotiation and building trust can improve the governance of biological invasions; addressing trade-offs and improving policy integration is a political process built on negotiation across stakeholders with different interests, values, and perspectives, which requires trust, ownership of the process, and learning (**section 6.4**).

Table 6 7 **Tactics to enable policy reform for invasive alien species policy.**

	Tactic	Expected benefit	Key references
1	Multidisciplinary to transdisciplinary research on invasive alien species policy regimes	Building robust and long-term resilience and the adaptive capacity of the governance systems for invasive alien species	(Daviter, 2017; Herrick, 2019)
2	Policy narratives to deal with the full continuous spectrum of service delivery or regulation, from prevention, eradication, control and restoration rather than treating each in isolation	Better integrated and effective policy regimes	(Daviter, 2017; Herrick, 2019)

Table 6 7

	Tactic	Expected benefit	Key references
3	Drawing on the full suite of adaptive and control-focused instruments as relevant **(Figure 6.8)**, including a combination of voluntary measures with regulatory and legislative frameworks	More comprehensive, effective and efficient governance for biological invasions	(Herrick, 2019; Primmer *et al.*, 2015; Shine *et al.*, 2000; Termeer *et al.*, 2010)
4	Inter-agency coordination to co-ordinate across policies and agencies and to monitor (stakeholder and Indigenous Peoples and local communities-wide or) government-wide activity	Increased efficiency and effectiveness of resource allocation and knowledge sharing	(Daviter, 2017; Herrick, 2019)
5	High autonomy for decision makers combined with strong, coherent, overarching policy	More efficient and targeted local solutions	(Vaas *et al.*, 2017)
6	Strategic and programmatic coordination that has adequate resourcing and authority to enable coordination	Improved effectiveness of implementation measures	(Daviter, 2017; Herrick, 2019)
7	Work towards intergenerational sustainability for invasive alien species by linking the consideration of ecosystem functions and process with management actions	Achieving environmental sustainability and political, stakeholder and Indigenous Peoples and local communities support	(Smolarz *et al.*, 2016)
8	Knowledge systems that enable sharing of information, concepts and arrangements across all stakeholders and scales	Improving learning to empower all stakeholders and Indigenous Peoples and local communities to manage invasive alien species	(Smolarz *et al.*, 2016; Staples & Hermes, 2012)
9	Creating space for multiple knowledge systems and experiences to encourage the recognition of different values	Building trust and social capital for effective collaboration and cooperation (collective action)	(Leventon *et al.*, 2021; McKiernan, 2018)
10	Implementing mechanisms for reviewing and monitoring policy effectiveness, including gathering data for "Response" indicators (following the Theory of Change, **section 6.2.1**) so that the success of management interventions can be assessed and fed into adaptive planning	Overcoming slow and inadequate implementation of policy	(McGeoch *et al.*, 2010; McGeoch & Jetz, 2019; OECD, 2019)
11	Integrating invasive alien species considerations into policies related to other environmental threats, including climate change	Policy that recognizes the inherent inter-dependencies of multiple forms of environmental change	(Smolarz *et al.*, 2016)
12	A focus and research on the relationships between policy instruments within and between sectors to determine what invasive alien species -relevant policy gaps exist, where policy conflicts occur, and how new policy can best complement existing policy – as the basis for a transition to integrated governance for biological invasions	Policy coherence, filling policy gaps and avoiding perverse incentives	(Visseren-Hamakers, 2015)

6.3 LEGAL AND REGULATORY OPTIONS

A broad array of international and national legal and regulatory instruments that directly or indirectly reference invasive alien species exist **(Table 6.8)**. These instruments aim to manage invasive alien species by preventing their introduction and spread and mitigating their impacts. They provide the formal rules upon which other policy instruments (e.g., economic, social; **Table 6.1**) can be framed and operated, and are also associated with multiple global, regional and national organizations. These instruments regulate or propose voluntary standards for the activities of different sectors (e.g., environment, production, extraction, health, trade and transport), often at different stages of the invasion process. This division of sectors, organizations, geopolitical scales and management by invasion stages highlights some of the main governance challenges of managing biological invasions discussed in **section 6.2**.

This section presents a suite of possible policy instruments to address the drivers and impacts of invasive alien species from a sectorial (**section 6.3.1**), geopolitical (**section 6.3.2**) and national (**section 6.3.3**) perspective. The options presented are brought together in **section 6.7**, where the need for alignment and coordination between legal and regulatory instruments across sectors, geopolitical scales and invasion stages is described. Meeting these needs would solve the current significant gaps in coverage of regulations and/or standards targeting invasive alien species and help to implement integrated governance based on sharing efforts and commitment and understanding the specific role of all actors (related to the principle of shared but differentiated responsibility). A key finding of the present assessment is that there is a need for coordination between policy initiatives to promote free trade, protect animal, plant and human health, or address other drivers of biodiversity loss, such as climate change and land-use and sea-use change. The section also shows options to improve the efficiency and effectiveness of invasive alien species intervention efforts at the national level and their integration at a regional scale.

6.3.1 Legal and regulatory options at and across sectors

Legal and regulatory instruments from many interacting sectors deal with management of biological invasions, either directly or indirectly. This section presents and discusses the legal and regulatory instruments aimed at solving some of the main challenges in the five key sectors described in **Table 6.8**. Rather than describing specific sector-by-sector solutions, legal and regulatory instruments that apply, in many cases, to more than one sector are presented. The options described here focus on addressing four main governance challenges:

- fragmentation across sectors,
- externalities,
- conflicting interests and trade-offs, and
- hurdles to policy implementation.

Table 6 8 **Some of the international legal, regulatory and organization-based instruments relevant to invasive alien species by sector.**

Adapted from Burgiel (2015).

Sector	Activities	Examples of relevant legal and regulatory instrument	Type of instrument*
Biodiversity and environment	Conservation and natural resource management	CBD	Binding
		Ramsar Convention	Voluntary
		Bern Convention	Binding
		United Nations Framework Convention on Climate Change (UNFCCC)	Voluntary
		Protocol on Environmental Protection to the Antarctic Treaty (the Madrid Protocol)	Binding
Transport and trade	Movement of goods, sanitary and phytosanitary measures and border security	WTO	Binding
		IMO	Binding
		International Civil Aviation Organization (ICAO)	Voluntary
		IPPC	Binding
		WOAH	Binding
		World Customs Organization (WCO)	Voluntary
Production systems	Agriculture (silviculture, horticulture, livestock husbandry), aquaculture, and Living Modified Organisms[6]	IPPC	Binding
		WOAH	Binding
		FAO	Voluntary
		Cartagena Protocol on Biosafety	Binding
Extraction systems	Forestry and fisheries	FAO	Voluntary
Public health	Protection against public health threats	WHO	Voluntary
		One Health Joint Plan of Action	Voluntary

* Binding instrument refers to those where signatories have a legal obligation to implement and/or achieve their commitments

6. Living modified organisms are any living organism that possesses a novel combination of genetic material obtained through the use of modern biotechnology as defined by the Cartagena Protocol (Bail *et al.*, 2014).

6.3.1.1 Addressing fragmentation challenges

Building on the interconnections between different sectors (e.g., transport, human health, trade, agriculture and aquaculture, forestry and biodiversity) to overcome policy fragmentation would provide a pathway for the effective prevention of invasive alien species (**Figure 6.10**). Such a pathway would benefit from a coordinated view of biosecurity across relevant agencies, and a clear definition of the roles and responsibilities of relevant national offices.

(1) Develop a coordinated approach to biosecurity across relevant agencies

A coordinated approach to biosecurity may help facilitate the export of products that otherwise would be subject to import restrictions in other countries. At the same time, it could protect agriculture, forestry, horticulture, fisheries, native biodiversity and human health. A coordinated view of biosecurity would mean blurring the lines between strong sectorial identities associated with specific international standards, individual economic sectors such as health, agriculture and the environment, specific research communities, and unique stakeholder and Indigenous Peoples and local communities involvement. Biosecurity can benefit from close collaboration between the various national agencies that oversee human health, trade policy, agriculture and aquaculture, forestry and biodiversity (CBD, 2012, 2018).

Efforts in the direction of cross-agency coordination have been proposed in reviews of existing biosecurity arrangements; for example, Australia's quarantine and biosecurity arrangements (CSIRO, 2022; Durant & Faunce, 2018) are at the core of the Great Britain Non-Native Species Strategy and its Secretariat (**Box 6.6**). A broader coordinated biosecurity approach can be achieved through close dialogue between health, agriculture and environment sectors; global, national and local authorities; and natural and social sciences. For example, the One Biosecurity approach (Hulme, 2020) provides a framework to tackle multiple social and environmental challenges: climate change, increasing urbanization, agricultural intensification, human global mobility, loss of technical capability as well as public resistance to pesticides and vaccines. This framework can benefit policy development regardless of the type of invasive alien species

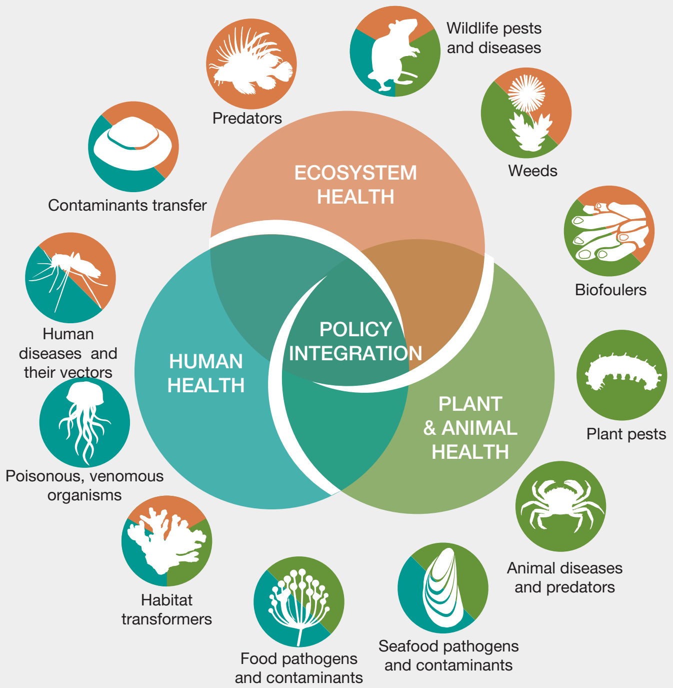

Figure 6 10 **A conceptual diagram of the links between different types of invasive alien species, human, animal, plant and environmental health that arise from the impacts of invasive alien plants, animals and pathogens, as described by the One Biosecurity approach (Hulme, 2020).**

Different types of invasive alien species will have different drivers, pathways and impacts, but their management and policy development aimed at prevention can all benefit from explicit recognition of the inseparability of human, ecosystem and plant and animal health and coordinated engagement by sectorial agents operating within each of those spheres.

presenting threats to nature, nature's contributions to people and good quality of life **(Figure 6.10)**.

A coordinated approach to biosecurity can be improved by additional capacity, including personnel, expertise and equipment. National agencies concerned with biological invasions may be able to build on efforts by agricultural and trade ministries to incrementally improve their sanitary and phytosanitary measures and border control systems. There may also be opportunities to fast-track development of these systems and address any knowledge and policy gaps by making relevant information available when and where it is needed across different offices. In many cases, knowledge sharing and adapting the practices and methods of countries with more advanced biosecurity systems could be an effective strategy in countries lacking biosecurity protocols (Hulme, 2021) and **section 6.6**. Finally, there may be creative opportunities to tailor capacity-building resources and materials currently offered by groups such as the World Bank, the IPPC, the WOAH **(Table 6.8)**, the WCO, regional development banks and national or regional research and development organizations.

(2) Clearly define roles and responsibilities across existing national offices within legal and regulatory instruments

Risk assessments are the most common approach to prevent the introduction of potentially harmful alien species from imports (**Chapter 5**, **section 5.2.2.1.e**), and are an essential component of any legislation enforcing regulations of trade. In fact, as explicitly stated by the WTO SPS Agreement, countries have the legal right to take proportional measures affecting trade based on the application of scientific principles. Deciding which governmental authority is responsible for assessing the risks of specific imports is important for preventing unwanted introductions. A clear definition of the roles and responsibilities of all agencies involved in the prevention of alien species introductions can help to devise effective management strategies (Hewitt *et al.,* 2006). Moreover, these roles and responsibilities can be supported by legal and regulatory frameworks that allow governments to execute the assigned tasks (Hewitt *et al.,* 2006). The questions in **Box 6.6** can help to guide governments to identify the most appropriate authority, and to strengthen the tools and methods for decision-making (as discussed in **Chapter 5**, **section 5.2.2**). These decisions could focus on defining the appropriate level of protection, and trade-offs between good quality of life benefits of the import and potential impacts on biodiversity. It is important to highlight that although one authority might be considered responsible for assessing the potential impact of an import, continuous communication and coordination between government agencies could ensure that all possible risk dimensions are taken into consideration.

Box 6 6 Governance and management questions to guide decisions on import proposals for aquaculture, horticulture, or silviculture species.

Adapted from Hewitt *et al.* (2006).

1. Does the government permit the importation of alien species?
2. Will any new species imports be allowed for production purposes?
3. Has an adequate risk assessment been conducted to support the decision to import the new species and provisions for managing potential harm?
4. Under what national regulation(s) will the import of a new species occur?
5. Which government agencies are responsible for management of these regulations?
6. Will these new species be allowed for uncontrolled release, within controlled or quarantine facilities?
7. Will the responsibility for managed species (e.g., in aquaculture/horticultural) be different from wild (e.g., wild, feral, or released species) populations?
8. Who will be responsible for the importation (e.g., private individual, research agency/university, industry, or government)?
9. Under what legislative arrangements will release into either a managed facility or the natural environment occur?
10. Who will be responsible for managing the release (e.g., private individual, research agency/university, industry, or government)?
11. Are there appropriate monitoring systems in place to detect and manage accidental releases in the environment?
12. Could neighbouring jurisdictions potentially be affected, and if so, are communication pathways in place to manage the risk?
13. Will neighbouring countries be involved in the decision-making process?
14. Do emergency response measures exist, including identification of the responsible authorities, in case of unforeseen negative impacts?

6.3.1.2 Addressing the indirect costs of biological invasions to uninvolved third parties: externalities

The overarching goal of the policy options presented in this section is to incorporate the negative impacts of invasive alien species into the social or economic context responsible for their introduction and spread. Such an approach would align the private, environmental and social costs of invasive alien species, so that trade-focused decisions take into consideration the environmental needs of society. It would ensure that all responsible government agencies are involved in attributing associated costs and that prices carry all the relevant information. This option would benefit from clear delineation of the environmental jurisdiction of non-environmental multilateral agreements and defining liability and redress from the negative impacts of invasive alien species on nature, nature's contributions to people and good quality of life.

(1) Delineate the environmental jurisdiction of trade agreements, so that the mandates of multilateral environmental agreements are enforceable

The tug-of-war between the philosophical underpinnings of biodiversity centred (grounded on the precautionary approach) and trade-related multilateral agreements (grounded on the evidence of adverse effects of an introduction) creates a conflict between trade and the environment (Stilwell & Turk, 1999). One way to avoid such conflict in the context of biological invasions is for governments to proactively define the relationship between trade and environmental centred agreements when negotiating multilateral environmental agreements. As discussed in Stilwell & Turk (1999), defining this relationship should not rely on exemptions ("saving clauses") in multilateral environmental agreements. Rather, agreements would better aim to establish a mutually supportive relationship between trade and the environment. Determining when the provisions in one of these two sets of agreements should supersede the other would help to internalize the externality of alien species impacts. It would also enhance policy coherence between multilateral agreements on trade and the environment, making these mutually supportive in favour of sustainable development (OECD, 2020). Specifically, this could bring about balanced and effective multilateral agreements for this transboundary and global environmental problem without the fear of trade barriers being invoked (Stilwell & Turk, 1999). This clarity would also help preserve the integrity of the multilateral trading system, which is increasingly criticized for its tendency to override social and environmental policies (European Commission, 2021). Likewise, it would address the view of social and environmental agreements as attempting to override multilateral trading rules (European Commission, 2021). Overall, such policy integration would also reduce the tendency and need to resort to unilateral trade measures, which would result in a lack of coordination and collaboration across jurisdictions.

(2) Defining liability and redress from the negative impacts to biodiversity of invasive alien species in multilateral and national legal and regulatory instruments

Invasive alien species can be viewed as a form of "self-regenerating pollution" (De Klemm, 1996). A "legal personality" (**Glossary**) or entity could therefore be regarded as liable for the damages caused from their involvement in the introduction of an invasive alien species. Preamble 33 of European Union Regulation 1143/2014 affirms that Member States should impose effective, proportionate and dissuasive sanctions for infringements, considering the nature and gravity of the infringement, the principle of recovery of the costs and the polluter pays principle. The same legislation, at art. 21 on cost recovery, says that "in accordance with the polluter pays principle … Member States shall aim to recover the costs of the measures needed to prevent, minimize or mitigate the adverse impact of invasive alien species, including environmental and resources costs as well as the restoration cost." However, the idea of liability and reparation for the impacts of an invasive alien species is missing from many multilateral environmental agreements. One notable exception is the Bern Convention which makes a formal recommendation about liability. Another example is the Convention on Civil Liability for Damage Resulting from Activities Dangerous to the Environment in Europe (Council of Europe, 1993) that specifies liability for genetically modified organisms or micro-organisms that present a significant risk for humans, the environment, or property.

Different objectives and guiding principles across legal and regulatory instruments raise complex questions about how liability for biological invasions can be assigned; and how liability can be enforced under the current state of international law. The use of environmental liability directives such as the Principle of Polluter-pays (for example, EU Directive 2004/35/CE) or nuisance laws (Pidot, 2005) provides one pathway to incorporate liability and redress provisions into the current multilateral environmental agreements. In these cases, damages are recognized as any unwanted change in protected species and natural habitats, water resources and/or soils (**Chapter 5, section 5.3.2**): namely, negligence or intentional actions of legal persons or entities involved in activities resulting in "environmental damage". In the context of transnational impacts, the best approach is that national legal and regulatory frameworks reflect obligations under international law and emphasize transboundary cooperation and collaboration concerning management of biological invasions, including liability for harm.

Given the nature of biological invasions, enforcing environmental liability would require shifting the burden of proof from the prosecution to the defendant(s) (Kramer, 2005; Pidot, 2005). Under such a regime, the prosecution would only have to demonstrate objective facts about the presence of an invasive alien species associated with a given activity of a legal person(s) or entity(ies); then the defendant(s) would need to prove the resulting invasion was not the product of negligence (Secretariat of the CBD, 2001). These proofs would be provided by all parties that received some form of financial benefit from the transport, sale and/or introduction of the species liable for some part of the harm (Secretariat of the CBD, 2001). Reframing who should be the target of punitive proceedings has the potential to develop a culture of accountability and responsibility, focused on encouraging voluntary compliance and implementations of best practices, though it is important that punitive actions are maintained as a potential last resort (Kramer, 2005; Pidot, 2005).

6.3.1.3 Addressing conflicting interests and trade-offs

Balancing the interests of multiple sectors and activities can be achieved through the development of legal and regulatory instruments, and reduce inconsistencies and misalignment in the objectives of legal and regulatory instruments. This approach could remove perverse incentives and, for example, encourage the transition to native species (**Glossary**) in production systems, stop the promotion of alien species as a tool to reduce poverty and increase food security, and increase the awareness of invasive alien species problems in disaster relief and assistance programmes.

(1) Removal of perverse incentives in sector-specific legal and regulatory instruments

Legal and regulatory instruments that promote trade, agriculture and aquaculture, infrastructure management and tourism can also facilitate invasive alien species introductions (**Chapter 3**, **section 3.2.5**) and exacerbate their impacts on biodiversity (**Chapter 4**, **section 4.3**). The removal, phase out, or reform of these incentives harmful to biodiversity is one of the Kunming-Montreal Global Biodiversity Framework targets (Target 18). As discussed by Herrick (2019) and Lodge *et al.,* (2006) aligning economic, social and environmental goals is the first step towards resolving perverse incentives (CBD, 2011). Such alignment can be achieved through careful evaluation of the trade-offs between policies with well-intentioned objectives, for example those aiming to improve good quality of life and nature's contributions to people but that promote the use of invasive alien species to do so. **Figure 6.11** showcases some examples of such perverse incentives. There are existing policy guidelines relevant to this topic (such as the European Union Green Paper on the Reform of the Common Fisheries; Commission of the European Communities, 2009), and new ones could be developed that focus on addressing specific perverse incentives. The analytical and policy guidance tools developed by the Organisation for Economic Co-operation and Development (OECD; OECD *et al.,* 2007; Sovacool, 2017) and the United Nations Environment Programme (UNEP; Morgan, 2008; Sovacool, 2017) are also valuable tools to start evaluating and addressing the possible biodiversity impacts of current and future legal and regulatory instruments.

Two factors could be considered to remove perverse incentives associated with activities that contribute to biological invasions. First is the resistance to substantive reform. In many cases, removing or modifying a policy can raise legitimate concerns about the economic consequences and the political capital cost of such changes. This is exemplified by the criticism of several countries of CBD COP decision VI/23 (CBD, 2002). In their view, Guiding Principle 7 (which advises member states to "implement border controls and quarantine measures, for alien species… based on a risk analysis of the threats posed by alien species and their potential pathways of entry") could be used as a tool to implement disguised trade barriers, thereby contravening the WTO SPS Agreement. A second factor to consider is the scale (spatial and temporal) at which proposed changes could potentially have an impact. In many cases, policy changes that can prevent invasive alien species introductions or reduce possible invasion drivers will have a direct, short-term economic and social cost for local communities, although communities would benefit from such changes in the long term. Identifying, understanding and adequately responding to the possible short-term social impacts of activities that promote the use of invasive alien species is one of the most challenging aspects of reforming policy instruments.

In addition to the examples shown in **Figure 6.11**, the development of a carbon sequestration economy could facilitate introductions of alien species (**Chapter 3**, **section 3.2.5**). This could take place *via* tree plantations through initiatives like Reducing Emissions from Deforestation and forest Degradation (REDD; Harvey *et al.,* 2010), national and multilateral initiatives on the use of biomass for energy production (i.e., EU, 2018; Jonsson *et al.,* 2021), and other restoration strategies (Brundu *et al.,* 2020) involving invasive, or potentially invasive, alien species. Tree planting is at the core of many national and regional climate strategies (i.e., carbon neutrality commitment by the European Union, China, United States, South Africa, Japan, South Korea and Canada; Climate Action Tracker, 2020). However, the most frequently used species in forestry plantations are trees from the genera *Pinus, Eucalyptus* and *Acacia* species. Though these species have traits that make them suitable for relatively rapid afforestation, they are also potentially highly invasive (Doughty, 2000; Eldridge *et al.,*

(A) Agriculture

Chemical control of alien agricultural pests has generated great concern by virtue of its non-target impacts on species of conservation concern. Carson (1962) raised this issue in her book that helped launch the environmental movement, and use of other broad-spectrum insecticides has led to similar impacts.

(B) Aquaculture

There are several instances where unintentional introductions may be associated with cultured target species; one of these being introduction of *Sargassum muticum* (wire weed) with oyster imports.

(C) Public health

The use of broad-spectrum insecticides to control introduced mosquitoes that vector human diseases, such as DDT, affected many bird species, including those of conservation concern, through eggshell-thinning. It also exacerbates pesticide resistance.

(D) Forestry

The forest-based industries and/or the energy production sector can be impacted by biological invasions. An example is *Eucalyptus globulus* (Tasmanian blue gum), which has invaded many forestry adjacent sites in North-West Spain.

(E) Infrastructure management

Road construction and roadside maintenance have often fostered invasion by non-native plants. Increasing establishment of mobile and immobile artificial structures (e.g. ships, barges, coastal defences, artificial reefs, offshore platforms) and canals play major roles in transmission, establishment and spread of aquatic non-native species.

(F) Military facilities

Midway Atoll was the site of a U.S. Naval Air Station from 1940 through 1996. The invasive plant *Verbesina encelioides* (golden crownbeard) was first observed in 1955 but was maintained at low abundance by intensive road and landscape maintenance. A shift in the 1990s to greatly reduced mowing and herbicide use triggered massive growth of this plant to the detriment of nesting seabirds, especially albatrosses.

(G) Tourism

National actions to promote local tourism (e.g., artificial reefs) in marine protected areas. Brazil's Tourism Ministry launched a plan to sink 1200 scrapped ships, trains, and airplanes, most of them inside marine protected areas, supposing that they will promote diving tourism business. Artificial reefs, such as those that would be created by sinking those vehicles, are used by invasive alien species (e.g., *Tubastraea* spp. (sun corals)) as stepping-stones to natural reefs, causing ecological, social, and economic deterioration.

(H) Biomass

Prosopis spp. (mesquite) are a group of species native to Central and South America which have been introduced in Australia, Asia, and Dryland Africa for fuel wood, fodder, to improve soils and reduce erosion. After the demand for *Prosopis* crashed, many plantation were abandoned without management or eradication. As a result *Prosopis* has become a major problem, particularly in Africa. It has severely impacted traditional livelihoods of pastoral communities.

Figure 6 11 **Examples of perverse incentives where actions aimed at promoting an activity fail to take into account the existence of environmental externalities.**

Examples present cases where (a) agriculture (Carson, 1962; Herms & McCullough, 2011), (b) aquaculture (Engelen *et al.,* 2015), (c) public health (Carson, 1962; Walker *et al.,* 2003), (d) forestry (Calviño-Cancela & Rubido-Bará, 2013), (e) infrastructure management (Galil *et al.,* 2017; Mineur *et al.,* 2012; Skultety & Matthews, 2017), (f) military facility management (Taylor *et al.,* 2020), (g) tourism (Miranda *et al.,* 2020) and (h) biofuels (Pasiecznik, 1999) can promote biological invasions. Photo credits: (a) James H. Miller, USDA Forest Service, Bugwood.org – under license CC BY 3.0 US / (b) Graça Gaspar, WM Commons – CC BY-SA 3.0 / (c) LSIS Helen Frank, WM Commons – Public domain / (d) Ignacio Amigo – CC BY 4.0 / (e) Rept0n1x – Walk to Lunt (102), WM Commons – CC BY-SA 2.0 / (f) Forest & Kim Starr, WM Commons – CC BY 3.0 US / (g) Paula Raposo – CC BY 4.0 / (h) Thamizhpparithi Maari, WM Commons – CC BY-SA 3.0.

1994; D. M. Richardson & Rejmanek, 2004). While not invasive, *Elaeis guineensis* (African oil palm) plantations have been promoted as a climate mitigation strategy, yet they have limited biodiversity and conservation value (Harvey *et al.,* 2010). Therefore, not considering fundamental environmental values, including safeguarding biodiversity in climate mitigation initiatives, can result in serious negative ecological consequences. Examples of impact include biotic homogenization (**Glossary**, Olden *et al.,* 2004), genetic swamping (R. C. Barbour *et al.,* 2010) and altered ecosystem processes (Simberloff *et al.,* 2009). Moreover, the escape of these plantation species can become costly to manage and lead to significant biodiversity losses (D. M. Richardson & Rejmanek, 2004).

(2) Encourage the transition to native species in production systems

Reducing the dependence of the still growing aquaculture, horticultural and silvicultural sectors on alien species is one of the most pragmatic approaches to reduce translocations of problem species within and across national borders. There are options to replace some cultivated alien species with native species (e.g., Jones Jr & Foote, 1991; Pérez *et al.,* 2003; van Heezik *et al.,* 2012). Mainstreaming this perspective can be done through technical and policy developments that promote potential candidate native species that are preferred by their respective communities and local consumers. Nonetheless, it is important to highlight that alien species constitute as much as 75 per cent of the species used for consumption (Palacios, 1997), and that these constitute the cornerstone for the economic activities of multiple communities. Therefore, such transition needs to consider the possibility of alien species replacement to fulfil food security and economic needs, with fisheries and forestry species more frequently invasive than terrestrial food crops (De Silva *et al.,* 2009; FAO, 2019).

Some alien species used in many production systems are causing significant losses in performance, primarily from inbreeding (E. O. Wilson, 1999). Rather than replenishing the stocks of these alien species with fresh germplasm obtained from their natural range, policy and voluntary codes of practice could promote the development of viable and profitable culture techniques for suitably selected native species. The pangasid fish culture (*Pangasianodon hypophthalmus* (sutchi catfish)) in the Mekong Delta is a successful example of the viability of gradually reducing the dependence on alien species *via* the replacement by native species (De Silva *et al.,* 2009; Nguyen, 2007). The aquaculture production of *Piaractus mesopotamicus* (small-scales pacu) in Argentina is another example of the successful use of native species to address decreases in capture fisheries (Quirós, 1990). *Piaractus mesopotamicus* production is now the second largest aquaculture species-based production in Argentina (FAO, 2016). However, success is mixed across examples of shifts to native species in production systems, with shifts not being feasible in aquaculture in Indonesia, Malaysia and Thailand, but underway in some cases in India and Bangladesh (De Silva *et al.,* 2006, 2009). Similar contrasting trends have been reported in aquaculture in Europe (Turchini & De Silva, 2008).

In the case of silviculture, options for the replacement of invasive alien with native species seem limited in commercial forestry with the most productive forestry species being alien (D. M. Richardson, 1998; D. M. Richardson & Higgins, 1998). In the Galapagos, where other invasive alien species (*Centrolobium paraense, Juglans neotropica* (andean walnut), *Swietenia macrophylla* (big leaved mahogany) and *Tectona grandis* (teak)) are established, conservation authorities are encouraging the replacement of invasive alien timber species with non-invasive alien timber species (*Cedrela odorata* (Spanish cedar) and *Cordia alliodora* (Ecuador laurel)) and horticultural species (*Psidium guajava* (guava), *Cinchona pubescens* (quinine tree); Richardson, 1998).

(3) Stop the promotion of alien species as a tool to reduce poverty and increase food security

Alien species are the cornerstone of many aquacultural practices aimed at improving food security. Examples are alien tilapia (*Coptodon* spp., *Oreochromis* spp. and *Sarotherodon* spp.), salmonids (*Salmo trutta* and *Oncorhynchus mykiss*) and oysters (*Crassostrea* spp., *Ostrea* spp. (flat oyster), *Argopecten* spp.; De Silva, 2012; Paini *et al.,* 2016; McBeath & McBeath, 2010). This also been the case for silvi/agro-cultural initiatives aimed at reducing poverty, where species such as eucalypts and *Leucaena leucocephala* have been introduced in Southeast Asia. Grasses and legumes from Australia, South Africa and North America have been introduced in experimental farms near Santa Cruz (Bolivia) and *Prosopis juliflora* (mesquite) has been introduced in Africa **(Figure 6.11)** for fuel-wood for the rural poor (Murphy & Cheesman, 2006; **Chapter 4**, **Box 4.9**). These species can escape production environments and have adverse impacts on biodiversity and ecosystems (**Chapter 3**, **sections 3.2.5** and **3.3.1.1**). A first step to avoid this problem could be a strategy shift by development assistance organizations to embed the preferred use of native species into their codes of practices. Exploring viable native alternatives as primary species for human consumption or as animal feed is an option for such organizations. This shift will benefit from being coupled with national policy that promotes a culture of native species cultivation valued by Indigenous Peoples and local communities.

However, the implementation of a “native species”-centred approach would require careful consideration in each case. For example, it is critical to consider if native species can provide viable alternatives to assure food security for ever increasing human populations, especially in rural populations of developing countries (De Silva *et al.,* 2009; Murphy & Cheesman, 2006; Shackleton, Richardson, *et al.,* 2019). It would be important to consider socioeconomic conditions, as well as the views and needs of Indigenous Peoples and local communities, so that good quality of life is not adversely affected. This is clearly the case for alien tilapias (*Coptodon* spp., *Oreochromis* spp. and *Sarotherodon* spp.) in China, Indonesia, the Philippines and Sri Lanka, where they play a major role in subsistence aquaculture systems by providing a relatively cheap source of animal protein, as well as considerable export income (De Silva *et al.,* 2004, 2009); simply put, shifting to a less productive native species may not be feasible.

(4) Develop guidance documents and codes of practice to reduce pathway risks for disaster relief and assistance programmes

The lack of awareness of the possible problems caused by invasive alien species in international development programmes has made disaster relief and assistance an major invasion pathway (Murphy & Cheesman, 2006; **Chapter 3**, **section 3.2.2.2**). Specifically, the invasive alien species problems created by international disaster relief and assistance programmes arise unintentionally from activities inspired by humanitarian motives. Many anecdotal reports (summarized by Murphy & Cheesman, 2006) link invasive alien species to these programmes (**Chapter 3**, **section 3.2.2.2**). As the negative effects of alien species (un)intentionally introduced *via* this pathway can be long lasting and outweigh positive impacts, increasing this awareness is important.

The most efficient line of defence to identify possible invasive alien species in aid packages might be developing biosecurity protocols, voluntary codes of practice and risk assessment protocols focused on preventing (un) intentional introduction in these programmes. Addressing the problems that have resulted from invasive alien species (un)intentionally introduced by past international assistance and aid /relief activities could be a priority for all associated stakeholders. However, conflicts of interest may arise where an environmentally damaging species is contributing to local livelihoods (as reviewed in Shackleton, Shackleton, *et al.,* 2019; **Chapter 5**, **section 5.6.1.2**). Therefore, careful consideration is necessary of the trade-offs between reductions in nature and good quality of life benefits due to invasive alien species impacts and the immediate needs of vulnerable populations.

6.3.1.4 Addressing implementation challenges in key areas

Effective management of biological invasions can be promoted in multilateral and national legal and regulatory instruments by including early warning systems (**Chapter 5**, **section 5.5.2**) in all environmental legal and regulatory instruments; considering all alien species as possible environmentally hazardous living organisms; developing strategies to regulate cross-border e-commerce (**Glossary**); increasing awareness of and improved compliance with (voluntary) codes of practices; and incorporating prevention and control of invasive alien species into protected areas and island management plans.

(1) Include early warning systems for invasive alien species into multilateral environmental agreements and national legal and regulatory instruments

Although biodiversity-related policy instruments dealing with invasive alien species have generic surveillance provisions (e.g., the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization to the CBD, Article 29 and subsequent decisions; Cartagena Protocol, Article 17; International Treaty on Plant Genetic Resources for Food and Agriculture Article 17.2, or the Ramsar Convention on Wetlands, Articles 3(2), 4(1), 4(3), 6(2) (InforMEA, 2021)), no instrument has a direct requirement for monitoring or the development of early warning systems. The need to develop effective global early warning and rapid response systems has been stated as a priority action by the CBD COP (decision IX/4 "In-depth review of ongoing work on alien species that threaten ecosystems, habitats, or species" and by the "Charter of Syracuse" on biodiversity, adopted at the G8 Environment Ministers Meeting (22–24 April 2009, Syracuse, Italy). Early warning and rapid response systems for invasive alien species can detect the occurrence of new invasive alien species, supported by activities to identify new species correctly and acquire all related information (European Environment Agency, 2010b).

The lack of clear provisions on early warning systems is due to the need for cross-national integration of effort and standards for effective surveillance, which has proven difficult for biodiversity policy instruments (Hulme, 2021). A notable exception that incorporates monitoring provisions is the Bern Convention. Monitoring and early warning system provisions is also part of many management recommendation documents, such as guidelines by the IUCN (IUCN, 2000; Tye, 2018) and the Council of Europe (Genovesi & Monaco, 2013). Monitoring and early warning system provisions could be incorporated into multilateral environmental agreements and national policy instruments. For these provisions to be effective, they could encourage participation across government and non-government agencies, the scientific community and the general public. Such engagement could, for example, be *via* a complaint system to a dedicated national authority, which has proven to be a successful monitoring tool for the Bern Convention (C. L. Díaz, 2010). Also, similarly to the reporting mechanism under the UNFCCC, periodic reporting on the numbers and identity of invasive alien species by member states could be part of the provisions of multilateral environmental agreements. An international biosecurity convention organization (similar to the secretariat of the Convention on the Conservation of Migratory Species of Wild Animals (CMS)), if established, could implement a global surveillance and monitoring network to provide early warning of new threats (Hulme, 2021). Creating an international warning system is a large task, but there is already ongoing research in this direction (Latombe *et al.,* 2017; Pagad *et al.,* 2018; Essl, Lenzner, *et al.,* 2020).

(2) Inclusion of all invasive alien species as environmentally hazardous living organisms

Three areas central to biosecurity are: the management of risks associated with the accidental introduction of pests and diseases with food and agriculture, the introduction and release of genetically modified organisms and their products, and the deliberate introduction and management of invasive alien species and genotypes (FAO, 2003; **Chapter 3**, **sections 3.3.1** and **3.3.5.2**). Many biosecurity protocols do not cover "hitchhikers" or contaminants (e.g., spiders in produce, ants in taro plants), making the case for a broader definition of hazardous organisms, one that includes possible alien species that may affect the environment, the economy and/or human health (such as the New Zealand Import Health Standards that include provisions for hitchhikers and contaminants).

In addition to a broad definition of "hazardous organisms" in trade-related biosecurity protocols, clear and accurate labelling for consignments of all living organisms being transported is needed to prevent invasion. Labelling can be supported by regulated species lists with prohibited (strict approach) or permitted (lenient approach) species listed by jurisdiction and supported by import risk analyses. This protocol could be accompanied by handling protocols that promote environmental safety during transport. Adequate labelling needs to be developed and implemented through an efficient exchange of information between vendors and national authorities. The WCO has developed cross-border e-commerce frameworks (WCO, 2018) and technical standards (WCO, 2019) for this purpose, focusing on the need for electronic data to manage the risks of cross border movements of goods, and has been asked at COP15 of the CBD to look specifically at the question of e-commerce and invasive alien species (CBD, 2022c). Through the exchange of advanced electronic data, and considering all traded species as hazardous organisms, national authorities can ensure compliance with regulatory requirements. Furthermore, the use of detailed and accurate labelling of consignments would make exporters active participants in biosecurity and responsible for clean trade (**Glossary**, Hulme, 2021).

The UN Economic and Social Council's Sub-Committee of Experts on the Transport of Dangerous Goods will consider including environmentally hazardous living organisms in chapter 2.9, class 9, of the United Nations *Recommendations on the Transport of Dangerous Goods – Model Regulations*,[7] at its upcoming session in 2023, taking into account the risk of unintentional introduction of invasive alien species, including pathogens. This could be a significant step forward if governments are willing to accept it.

(3) Develop and raise awareness of codes of practices and standards and other mechanisms to regulate cross-border e-commerce

The rise of e-commerce contributes to invasive alien species spread (**Chapter 3**, **section 3.2.3.1**), and is becoming an increasingly critical biosecurity concern (Ricciardi *et al.,* 2017; **Chapter 5**, **section 5.3.1.1**). However, the online trade of living organisms is poorly regulated (Lenda *et al.,* 2014; Mazza *et al.,* 2015). Also, the high level of anonymity in online trade can circumvent accountability and taxes. Individuals and small companies that sell through the internet may not be legally registered and often do not disclose their specific location of operation. As a result, consignments of regulated articles can be imported into a country without any effort to meet the phytosanitary requirements of the receiving country (Derraik & Phillips, 2010; Keller & Lodge, 2007; Morrisey *et al.,* 2011). Buyers and sellers in the plant and animal trade may be ignorant or misinformed on potential dangers and biosecurity regulations, or may incorrectly identify their products (Giltrap *et al.,* 2009; Walters *et al.,* 2006). As such, national lists of regulated species are more important than ever. For example, the European Union, through the European Union Regulation 1143/2014, has adopted a List of species of Union Concern that are banned from import into the Union, and for which there is a general obligation to eradicate or control when recorded in the territory of a member state. The list is regularly updated. In 2004, Japan adopted an Invasive Alien Species Act that includes a list of regulated species.

Based on these lists, voluntary codes of conduct for e-commerce platforms could be developed as cost-effective approaches to address the trade-off between the economic benefits of e-commerce and the risk of environmental harm from the scope of these species (Monaco & IUCN Invasive Species Specialist Group, 2021; Shackleton, Adriaens, *et al.,* 2019). Specifically, the adoption of voluntary codes of conduct could prevent sales and auctions of species into countries where they are regulated and improve correct labelling of traded species. These conduct codes could

7. United Nations publication, Sales No. E.19.VIII.1.

stimulate e-commerce platforms to self-regulate by screening their own listings for species of concern and proactively complying with countries' invasive alien species laws, requiring sellers on online platforms to provide information on the species they sell. At a minimum, this information would include taxonomy, a record of potential invasiveness of these species, and appropriate measures that a buyer could use to prevent a species escape or release. Clear labelling of consignments, combined with lists of prohibited or permitted organisms, is perhaps the best way to prevent environmental harm without imposing major constraints on e-trade.

Such voluntary guidelines are in the process of being implemented for endangered and threatened species trade by the world's leading technology companies, including e-commerce and social media companies (e.g., Alibaba, eBay, Facebook, Google, Instagram, Microsoft, Pinterest, etc.). One example of a related effort is the work by the Coalition to End Wildlife Trafficking Online.[8] It is a partnership between environmental organizations such as Trade Records Analysis of Flora and Fauna in Commerce (TRAFFIC), World Wildlife Fund for Nature (WWF) and the International Fund for Animal Welfare (IFAW) with companies from across the globe to reduce wildlife trafficking online. Expanding these standards to include invasive alien species would help reduce introductions of invasive alien species through e-commerce.

The effectiveness of voluntary guidelines, codes of conduct and standards can be improved through the development of an efficient information exchange system accessible to all parties involved in trade and transport. At COP15, CBD parties considered measures on e-commerce and associated risks of invasive alien species (CBD, 2022c). One of these measures is considering the implementation of a single-entry system that facilitates the sharing of standardized information and documents with a single-entry point (i.e., a "Single Window approach") to fulfil all import, export and transit-related regulatory requirements.

(4) Increase awareness, participation and compliance with (voluntary) codes of practice for the translocation and exploitation of invasive alien species

Increasing governmental support for deregulation combined with industry opposition to restrictive legislation has led to a progressive emphasis on corporate responsibility and voluntary codes of practices worldwide (Sethi, 2011). Several activity-specific voluntary codes of practices have been developed to address the management of invasive plant species by the ornamental nursery industry (Baskin, 2002; Heywood & Brunel, 2009 and **Box 6.7**), aquaculture and forestry. Similar codes of conduct have been developed in Europe for several relevant activities, including boating, botanic gardens, hunting, international travel, pets, recreational fishing, zoological gardens and aquaria (such as the European code of conduct on hunting and invasive alien species, and other available at EASIN (2021)). These codes of practice provide practical and concise guidance in establishing common standards of good practice and responsible attitudes and behaviours for use of alien species in production activities. Their recommendations are intended to be complementary, not replace, the binding obligations embedded into national legislation and action plans that regulate any activity that transports, sells, or uses alien species.

The effectiveness of voluntary codes of conduct has limits but can be a valuable part of integrated systems to reduce the risk of invasive alien species. In fact, several codes of practice on invasive alien plants are in use throughout the world. Codes of practices on invasive alien species have two main goals: (1) to reduce deliberate introductions of invasive plants and (2) to increase the level of awareness-raising (Halford *et al.,* 2014). For example, the European and Mediterranean Plant Protection Organization (EPPO) standard PM 3/74(1) provides guidelines on the development of a Code of practice on horticulture and invasive alien plants (EPPO, 2009).

(5) Policy to support incorporation of management of biological invasions into protected area management plans

The designation of parcels of land- or seascapes as protected areas does not confer immunity from the effects of invasive alien species, and the invasion of protected areas erodes the maintenance of species diversity and natures contributions to people (e.g., terrestrial plants, Foxcroft *et al.,* 2013; Liu *et al.,* 2020; and marine systems, Gallardo *et al.,* 2017; Giakoumi *et al.,* 2019). This has become a concern at an international level, with many global conventions, policies or strategies focused on the threat of invasive alien species in protected areas in development (Foxcroft *et al.,* 2017; Shine *et al.,* 2000). The designation of protected areas will likely increase as a result of the Kunming-Montreal Global Biodiversity Framework adopted at COP15 of the CBD in late 2022, which clearly encouraged governments to increase protected areas on land and in water by restoring (Target 2) or protecting (Target 3) at least 30 per cent of terrestrial, inland water and of coastal and marine areas.

Due to the importance of protected areas, policy instruments could be developed to elevate protected areas to priority invasive alien species management sites, using a site-based management strategy (**Glossary**; **Chapter 5**, **section 5.3.1.3**). The development of such instruments may lead to the formalization of alien species management plans into the protected area management planning process.

8. https://www.endwildlifetraffickingOnline.org/

Box 6 7 **The way forward for ornamental horticulture and invasive alien plants: How to reduce risks and achieve sustainability?**

Ongoing innovation, cultivation and introduction of new plants has been considered critical to the survival and profitability of the horticultural sector, and can result in the ongoing introduction of alien species (Seaton *et al.,* 2014). Ornamental horticulture fosters plant invasions in a number of ways (Culley *et al.,* 2011). First, it often involves multiple introductions which include both the initial introduction as well as the subsequent sale and distribution of cultivated individuals through supply chain, retail centres, mail order catalogues, and over the internet (Culley *et al.,* 2011). Second, selective breeding may unintentionally favour traits, such as rapid growth, rapid seed germination, drought tolerance and disease resistance, that can enhance spread and invasive potential. Third, ornamental horticulture may promote invasiveness through commercialization of cultigens (plants known only in cultivation; Spencer & Cross, 2007). Although some may be self-sterile, different cultigens planted together may cross-pollinate and form viable fruit that is dispersed into natural areas. This has already been documented, for example, in *Pyrus calleryana* (callery pear) and *Lythrum salicaria* (purple loosestrife; **Figure 6.12**; Culley *et al.,* 2011; Culley & Hardiman, 2007, 2009).

To prevent further plant invasions resulting from ornamental horticulture, countries could commit to promote the inclusion of specific guidelines for the ornamental horticulture sector and supply chain, within the framework of national strategies on biological invasions and within related national policy (for example those relevant to biodiversity, SDGs, sustainable agriculture and forestry). As discussed by Hulme *et al.* (2018), closing this plant invasion pathway can be achieved by government-industry agreements to fund effective pre-and post-border weed risk assessments. This can be supported by widely adopted industry codes of practices. One example is the Code of Conduct for Invasive Alien Trees (Brundu & Richardson, 2016), which complements other European existing codes of practice dealing with horticulture and botanic gardens (Heywood & Sharrock, 2013). Codes of practice help producers and consumers make informed choices and help to target public education needed about horticultural invasion risks. For example, Green Lists of non-native ornamental species that have been assessed as having a low risk of escaping cultivation can contribute to the prevention of plant invasions (Dehnen-Schmutz, 2011)

Figure 6 12 **Left: *Pyrus calleryana* (callery pear). Right: *Lythrum salicaria* (purple loosestrife).**

Some cultigens planted together may cross-pollinate and form viable fruit that is dispersed into natural areas. Photo credits: Bruce Marlin, WM Commons – under license CC BY 3.0 (left) / GartenAkademie, WM Commons – under license CC BY 3.0 (right).

For example, the European Union Natura 2000 network aims to ensure the long-term survival of valuable and threatened species and habitats. Therefore, management plans are needed to prevent the deterioration of habitats and significant disturbance to species (Underwood *et al.,* 2020). The European Union invasive alien species regulations also provide legal support to prevent the introduction and spread of invasive alien species of Union concern, as Member States are obliged to undertake action to prevent and/or limit the impact of invasive alien species of Union concern (Underwood *et al.,* 2020). In Argentina, the national invasive alien species strategy requires management plans for the control of invasive alien species to be incorporated into protected area management and annual operational plans in national, provincial and municipal protected areas and private reserves (Paola & Kravetz, 2004). Protected areas in marine and connected systems are, however, less likely to achieve successful management following establishment of an invasive alien species (**Chapter 5**, **Figure 5.1**; Simberloff, 2021; **Box 6.8**). Therefore, high-level policy instruments are needed to focus on preventative measures, for example, ballast water management systems and biofouling protocols (**Table 6.4**; **Chapter 5**, **section 5.4.4.1**).

The application of high-level indicators downscaled from global to local levels may provide a framework for protected areas to assess progress in managing biological invasions. For example, in protected areas, it is possible to monitor the number of alien taxa, the species negatively impacting biodiversity, and the trends therein (**section 6.6.3**). This indicator can be read in conjunction with indicators such as trends in species at risk of localized extinction in a protected area. Response indicators provide protected areas and conservation agencies with feedback on the extent to which essential policy and management approaches have been adopted (McGeoch *et al.,* 2010; **section 6.6.3**). Collated at a national level these responses can be used to inform global indicators such as those for measuring progress in achieving the targets of the Kunming-Montreal Global Biodiversity Framework.

Box 6 8 **Marine protected areas as hotspots of invasive alien species.**

Marine protected areas are created to conserve the diversity of native species and associated habitat, and protect this biodiversity from threats such as invasive alien species (Francour *et al.,* 2010). Yet, evidence is emerging that suggests marine protected areas do not provide effective, or even adequate, resistance to the introduction, establishment and spread of invasive alien species (Usher *et al.,* 1988). For example, a survey of the venomous *Pterois* spp. (lionfishes) in 71 Caribbean reefs shows that they have established in high densities on reefs with depauperate native predator assemblages, and on reefs with both a high diversity and high biomass of native predators (Hackerott *et al.,* 2013; **Figure 6.13**). A census of lionfish in the Florida Keys National Marine Sanctuary, long considered a safe refuge for biodiversity (Hickerson *et al.,* 2012), revealed a rapid increase in their spread, abundance and biomass (Ruttenberg *et al.,* 2012). On the island of Martinique, despite intensive population control efforts (e.g., public awareness, authorize and equip dive centres for Lion-fish removal; Trégarot *et al.,* 2015) inspired by the Regional Caribbean Lionfish Strategy (Gómez Lozano *et al.,* 2013), lionfish colonized the west coast of Martinique, most of it designated as marine protected areas [9]. The lionfish also colonized the isolated Parque Nacional Arrecife Alacranes and the offshore coral reefs of the Flower Garden Banks National Marine Sanctuary in the Gulf of Mexico (Johnston *et al.,* 2017; López-Gómez *et al.,* 2014).

Figure 6 13 **Lionfish are now invading the western Atlantic Ocean, from North Carolina to Brazil.**

Photo credit: Oren Klein – under license CC BY 4.0

Box 6 8

In a global hotspot (**Glossary**) of marine invasive alien species, the Mediterranean Sea, a survey of rocky reef fish assemblages in 30 marine protected areas did not find evidence for any effect of marine protected areas on invasive alien species (Guidetti *et al.,* 2014). There is evidence of up to two times higher biomass of invasive alien fish in some marine protected areas than in adjacent unprotected areas (Giakoumi *et al.,* 2012), with up to 50 per cent of fish biomass in protected areas being invasive alien species (Giakoumi, Pey, *et al.,* 2019). An assessment of the vulnerability of 142 Mediterranean Sea marine protected areas to invasive fishes and algae found that Levantine marine protected areas are dominated by invasive alien species (D'Amen & Azzurro, 2020). Effective invasive alien species policy for marine protected areas should therefore include a strong focus on prevention, along with context-specific environmental management to minimize the suitability of local habitats to invasion as currently proposed in the Action Plan Concerning Species Introduction and Invasive Species in the Mediterranean Sea (UNEP, 2014).

6.3.2 Legal and regulatory options across geopolitical scales

This section presents and discusses strengthening legal and regulatory instruments at multilateral scales **(Table 6.4)**. The goal is to present the main strategies by which agreements, laws and regulatory instruments and voluntary codes relevant at broad geopolitical scales can be strengthened. The options presented focus on the need for clear national strategies, embedded in a regional context and framed by coordinated multilateral environmental agreements.

6.3.2.1 Link national invasive alien species strategies into regional plans to align efforts and complement national strategies

The drivers facilitating invasive alien species are transboundary in nature (**Chapter 3**, **sections 3.1.1** and **3.1.3**), and their impacts are rarely restricted within political boundaries (**Chapter 4**, **sections 4.3**, **4.4**, **4.5**, **4.6**). Actions by individual countries are therefore not enough to mitigate the drivers and address the impacts of invasive alien species. Therefore, effective responses are based on shared objectives, means and approaches across legal and regulatory instruments, while also being supported by cross national collaborative actions. The need for countries to work together to identify, share information on, and coordinate around common priorities on invasive alien species could be a priority in any future or renegotiation of regional agreements. Efforts in this direction are in place in multiple trading blocs like the Southern Common Market (MERCOSUR) and the European Union **(Box 6.9)**. Ensuring this coordination may be the most effective means of reducing the risk of new invasive alien species and further spread of invasive alien species. Moreover, the gains achieved by placing national policy priorities within the context of regional and international instruments outweigh the investment required for any country deciding to go "solo". This is clearly evidenced by the existing web of policy ties across geographic scales that transcend the invasive alien species issue.

9. http://campam.gcfi.org/CaribbeanMPA/mapview.php

Box 6 9 **From Regional policy to national priorities: MERCOSUR and European Union cases.**

MERCOSUR is a regional trade agreement whose members are Argentina, Brazil, Paraguay, Uruguay and Venezuela. These countries have agreed on several regional commitments promoting action on invasive alien species: MERCOSUR Biodiversity Declaration (2006); Article 7 of the MERCOSUR Environment Framework and the Work Plan of the MERCOSUR Working Group, which acknowledges the need for managing priority invasive alien species. Also, the MERCOSUR working subgroup on the environment (SGT6) acknowledged the need to define a joint work plan on invasive alien species, including prevention, control and eradication priority actions, as part of multilateral environmental agreements and the MERCOSUR Framework Agreement on the Environment (LXIV Ordinary Meeting, 2017). The adoption of risk assessment protocols for introduction of species and MERCOSUR Resolution GMC 38/2019 on Guidelines for the Prevention, Control and Mitigation of Invasive Alien Species to reduce impacts on biodiversity, environment, health, production, economy and culture, is one of the recent regional agreements (MERCOSUR, 2019).

These agreements have not been reflected in Actions plans or in practical measures to deal with invasive alien species in different MERCOSUR countries. Two exceptions are Argentina and Brazil, both of which developed a national strategy on invasive alien species (GEF, 2016; Ministério do Meio Ambiente/ Secretaria de Biodiversidade, 2018). Uruguay recently included the promotion of the control of invasive alien species as a general goal in their National Strategy of Biodiversity (Uruguay:

Box 6 9

MVOTMA-DINAMA, 2016) and, in Paraguay, the national policies of invasive alien species are related to Wild Life Law and Resolutions 1184-85 (SEAM, 2006), which deal with control of introduction of exotic fauna and flora species.

The European Union as a party to the CBD took actions to ensure its policies comply with Article 8h of the Convention, on invasive alien species. In 2014 the European Union agreed to a legislative text related to invasive alien species, adopted by the European Parliament and the European Council: Prevention and Management of Invasive Alien Species, European Union Regulation 1143/2014 (Council of the European Communities, 2014), fulfilling Action 16 of Target 5 of the European Union 2020 Biodiversity Strategy, as well as Aichi Target 9 of the Strategic Plan for Biodiversity 2011–2020 under the CBD (Baquero *et al.,* 2021). This adoption resulted in the immediate existence of an enforceable law for all Member States (January 2015). This Regulation also emphasizes prevention, early warning systems and rapid response, while also recognizing that when prevention fails, eradication is the best management alternative, alongside long-term control measures. In summary, Regulation 1143/2014 allowed overcoming the limited coordination of national strategies on invasive alien species assisted by an information system, the European Alien Species Information Network (EASIN)[10] including an early warning system supporting early detection of invasive alien species of Union concern in Europe. It stimulates strengthening of ecosystem resilience through restoration linking with other policies, e.g., the Marine Strategy Framework Directive and the Water Framework Directive to improve the control of aquatic invasive alien species in European Union countries (Boon *et al.,* 2020; Council of the European Communities, 2008). An interesting component of the European legislation is that it includes an obligation for Member States, at Art. 13, to enforce an action plan addressing relevant pathways of introduction of invasive alien species, thus focusing on prevention rather than on reaction to new invasive alien species.

6.3.2.2 Increase coordination across multilateral environmental agreements

Over the past decade, there has been widespread adoption of multilateral environmental agreements (**section 6.1.3**). However, this pool of multilateral legal and regulatory instruments is piecemeal in the way the pathways and impacts of invasive alien species are addressed. One way to overcome this is to consider a more comprehensive international approach that focuses on sustainable development *via* the protection of biodiversity and maintaining good quality of life. The Inter-agency Liaison Group on Invasive Alien Species established by the Executive Secretary of the CBD was central to this process, but has been inactive for several years. However, at COP15, Parties invited “the Secretariat of the United Nations Economic and Social Council, the World Customs Organization [WCO], the International Plant Protection Convention [IPPC], the World Organisation for Animal Health [WOAH], the World Health Organization [WHO], the Food and Agriculture Organization of the United Nations [FAO] and its Codex Alimentarius, the Secretariat of the Convention on International Trade in Endangered Species of Wild Fauna and Flora [CITES] and the Invasive Species Specialist Group [ISSG] of the International Union for Conservation of Nature [IUCN], within the scope of their respective mandates, to support the national implementation of the Kunming-Montreal Global Biodiversity Framework with regard to targets and actions related to invasive alien species, including their monitoring and reporting”(CBD, 2022c).

While discussions leading to an increase in coordination across multilateral instruments is more likely to take place for global scale instruments due to their global scale implications, coordination of regional multilateral instruments faces many difficulties and obstacles, particularly in developing countries. Linguistic, cultural and political differences within regions also pose major obstacles for coordination. As discussed by Burgiel (2015), the Caribbean region is an example of the difficulties of integrating regional policy instruments due to multitude of languages and historical affiliations, the status of overseas territories and political issues. That said, the efforts of regional intergovernmental or nongovernmental entities provide a primer for building synergies and increasing coordination across legal and regulatory instruments (Burgiel, 2015).

6.3.2.3 Embed in multilateral agreements mechanisms to enhance coordination and information exchange between policy instruments

The inherent complexity in managing biological invasions could be countered by communication and information exchange among a wide range of stakeholders including across national borders and across government agencies, the private sector, the scientific and research community and the general public. A system for communication and information exchange would be supported by a well-functioning infrastructure. Effective decision-making for alien species is only possible with timely access to scientific and technical information. Embedding the need of institutions that facilitate/mandate information generation and exchange into multilateral agreements can provide a mechanism

10. https://easin.jrc.ec.europa.eu/easin/

to achieve the goals of existing agreements and provide information for the application of guidance documents. At a minimum, identification and monitoring of alien species would be part of the mandate of the proposed knowledge-generating institutions. Most of the international legal instruments, agreements and texts relevant to invasive alien species highlight the importance of risk analysis principles, notification procedures and information exchange. Examples of information exchange infrastructures include the CBD Access and Benefit-sharing Clearing-House and the FAO Forest invasive species home. However, information is usually sectorized in instrument focused clearing houses. There is also a problem with limited exchange of information across government agencies, and from the scientific and research community to policymakers and the public (**section 6.6**).

The need for an organizing (multi)national authority is clear due to the fragmented nature of existing policy instruments, which limits the capacity of a coordinated and unified response to the invasive alien species problem (Shine *et al.,* 2000; Stoett, 2007). The benefits of creating such an organizing (multi)national authority are the capacity to:

- Coordinate policy across national agencies and countries within a trading block;
- Unify risk analysis of invasive alien species introduction pathways, and how these risk analyses are implemented by relevant authorities;
- Develop effective early detection and rapid-response activities;
- Promote the exchange of information among all the stakeholders and Indigenous Peoples and local communities involved in the prevention, control and eradication of invasive alien species;
- Enhance the capacity to be cost effective in preventing and mitigating strategies given the interdependence of management;
- Enhance the capacity to synthesize and integrate information from international agreements, regional/national agencies, sectorial initiatives and university research.

An example of such a multilateral coordinating body is the Committee for Environmental Protection (CEP) which has, for example, developed a Non-Native Species Manual for activities of the nations active in the Antarctic **(Box 6.10)**.

Box 6 10 **The Committee for Environmental Protection as a coordinating body for Antarctic alien species problems.**

Policy context: Policies that are relevant to biodiversity and to ecosystem services in the Antarctic region are developed, usually independently, by the Antarctic Treaty Consultative Parties (ATCPs), the Commission for the Conservation of Antarctic Living Resources (CCAMLR) and by the States responsible for the islands in the Southern Ocean (north of 60°S). The ATCPs are advised by the Committee for Environmental Protection (CEP), established by the Protocol on Environmental Protection to the Antarctic Treaty of 1991 and by the Scientific Committee on Antarctic Research (SCAR), a committee of the International Science Council (Protocol Article 10.2; **Supplementary material 6.1**).

Invasive alien species: Based on the advice of the CEP, the ATCPs have placed significant focus on preventing invasive alien species introductions to and impacts on the area south of 60°S. Current guidance for doing so is encapsulated in the Non-Native Species Manual of the CEP (ATCM, 2019), hereafter the Manual). The Manual covers the unintended introduction of species to the Antarctic region and the movement of species within Antarctica, and is an example of the effective translation of recent research to policy through the CEP (e.g., Hughes & Convey, 2010, 2012; Lee & Chown, 2011). Although the pace of such translation and uptake has been criticized and there is a lack of evidence to quantify the implementation of different biosecurity measures across more than a handful of national programmes, the rate of development of responses within the Antarctic Treaty System (ATS) has been relatively rapid, with these responses exceeding those typically expected elsewhere, as measured through a comparison with international responses to the relevant Aichi Targets of the Strategic Plan for Biodiversity 2011-2020 (Chown *et al.,* 2017).

Practical guidelines: The Manual has also been supplemented by other practical guidance for those operating in the region. Perhaps the best example is the COMNAP/SCAR Invasive alien Species Voluntary Checklists for Supply Chain Managers (SCAR & COMNAP, 2019), which provides practical guidance (and the evidence underlying it) to prevent the introduction of non-indigenous species to Antarctica. Other organizations, such as the Antarctic tourism industry body, the International Association of Antarctica Tour Operators (IAATO), have similar guidance for its members (IAATO, 2020). In the 2019/2020 season, more than 74,000 tourists visited the Antarctic and numbers are expected to rise.

Although the Manual makes reference to marine invasions, including the Practical Guidelines on Ballast Water Exchange in the Antarctic Treaty Area (ATCM, 2006), it also identifies the need for further guidelines for preventing and responding to marine invasive alien species (McCarthy *et al.,* 2019). The

Box 6 10

Manual contains a great deal of advice for terrestrial systems, including flow charts on how to respond to introductions. Notwithstanding all of the advice and agreements, Antarctic Treaty policy implementation proceeds through implementation in national law, which is highly variable between the nations that are party to the Antarctic Treaty and Protocol (Hughes & Pertierra, 2016). These include all nations that are active in Antarctica.

Progress and prospects: What should be done to limit the impacts of invasive alien species and the reasons for doing so, are uniformly articulated to the ATCPs *via* the Manual (ATCM), 2019). As a result, considerable progress has therefore been made in addressing the requirements for reducing the introduction and spread of invasive alien species, in monitoring the situation, and in responding to new incursions and developing eradication approaches (Hughes & Convey, 2012; McGeoch *et al.,* 2015).

The broader Antarctic region is changing rapidly as a consequence of global climate change (Le Roux & McGeoch, 2008; Lebouvier *et al.,* 2011; Rintoul *et al.,* 2018; Swart *et al.,* 2018), with most analyses indicating that risks of establishment, spread and impact of alien species will increase (Frenot *et al.,* 2005; Aronson *et al.,* 2015; Duffy *et al.,* 2017; McClelland *et al.,* 2018; McCarthy *et al.,* 2019; Pertierra *et al.,* 2020). Human activity in the region is also growing due to growth in scientific stations and numbers of science and support personnel, and in numbers of tourists (Chown & Brooks, 2019). Thus, invasive alien species policy requirements for the future will have to focus especially on what these changes mean for introductions from elsewhere into the Antarctic region. In the face of these challenges, a focus on better and coordinated biosecurity measures, for prevention, and the development of clear surveillance policy and practices to identify and characterize new establishments as they occur is essential, especially for marine systems (Aronson *et al.,* 2015; Hughes *et al.,* 2015; Hughes & Pertierra, 2016).

6.3.3 Legal and regulatory options at national scales

National investment in invasive alien species prevention and control generally requires governments to take the lead, especially if outcomes are intended to fulfil the public interest (Early *et al.,* 2016). As discussed above, a coordinated approach can indeed be challenging for several reasons, including administrative fragmentation and the need to take into account free trade agreements. To take the lead, each level of government needs the legal mandate to develop its relevant invasive alien species strategy (e.g., Genovesi & Shine, 2004) and collectively agreed implementation plan (e.g., National Invasive Species Council, 2008). In addition to these plans, countries could adopt legal and regulatory options to address invasive alien species, such as regulation of import; regulation of possession, trade, transport and reproduction in captivity; regulation of introduction into the wild; mandatory management of pathways of introduction; or mandatory eradication or control actions.

As with all significant government-led investments, legislative authority is necessary to engage budgetary expenditure (V. M. Adams *et al.,* 2018). Quarantine, biosecurity, environmental protection or marine protection acts of government can form the legislative basis for spending (Genovesi *et al.,* 2015; Pyšek *et al.,* 2020). Once legislative authority is in place then national invasive alien species strategies and management plans can be developed and implemented through collective decision-making and a co-investment process following (as far as possible) best practices (e.g., Victorian Government, 2010). National scale benefits from management of biological invasions cannot happen without this legislative authority empowering a government led response.

6.3.3.1 National invasive alien species strategies that identify the full suite of policy and management needs and priorities

Except for some provisions under the Sanitary and Phytosanitary Agreement, most governments have done relatively little to establish policies and programmes intended to limit the movement of invasive alien species (**section 6.1.3**; Early *et al.,* 2016; Pagad *et al.,* 2020; Turbelin *et al.,* 2017). Specifically, relatively few countries have invested in a comprehensive "biosecurity" approach or coordinated policies and programmes across relevant sectors for the management of biological invasions (**section 6.3.1**). One of the most effective and comprehensive approaches that governments can implement to minimize the spread and impact of invasive alien species is the development of national strategies and associated action plans, such as National Invasive Species Strategies and Action Plans (NISSAPs). Strategies for preventing and controlling invasive alien species can also be incorporated into NBSAPs to ensure policy connections and coherence. NBSAPs could also be used to enhance the coordination between other sectors, as suggested in **section 6.3.1.1**, *via* a national office as has been the case for the Great Britain Invasive Non-Native Species Strategy **(Box 6.11)**.

Following the recommendations and considerations included in multiple CBD decisions, strategy and action plans could:

- Define clear and measurable national targets (CBD decision X/2 paragraphs 3(b) and (c)).
- Mainstream biodiversity, communication, monitoring and reporting (CBD decisions X/2 paragraphs 3(d) and (f); XI/8 paragraph 4) into broader environmental, economic and social national and local plans (CBD decisions X/2 paragraph 3(d), X/33 paragraph 8(k)).
- Define funding needs (CBD decisions X/31 paragraph 11 and XI/ paragraph 17).
- Have a clear mechanism for providing financial resources (CBD decisions X/26 paragraph 3 and XI/ paragraph 17, 25) and ensuring resource mobilization (CBD decision X/3 paragraph 2).
- Promote cooperation with other multilateral environmental agreements (CBD decisions X/5 paragraph 3 and XI/6 paragraphs 10 and 11), particularly those focused on addressing climate change (X/33 paragraph 8(k) and XI/19 paragraph 7(a)).
- Have specific considerations for protected areas (CBD decisions X/31 paragraphs 1(c), 11, 26 and XI/2 paragraph 1(a)).
- Have specific considerations for different ecosystems such as marine/coastal (CBD decision X/29 paragraphs 7, 18, 67), islands (CBD decision X/15 paragraph 4(b)), mountains (CBD decision X/30 paragraphs 4, 8) and dry and sub-humid lands (CBD decision X/35 paragraph 2(g)).
- Broadly consider the value of nature by engaging multiple sectors (CBD decisions X/32 paragraph 2(g) and X/44 paragraph 6).
- Provide positive incentives (direct or indirect) that encourage achievement of biodiversity-friendly outcomes or support activities that promote the conservation and sustainable use of biodiversity (CBD decisions X/44 paragraph 6 and XI/30 paragraphs 3, 6, 7, 9).
- Consider gender dimensions (CBD decisions X/19 paragraph 5 and XI/9 paragraph 7), Children and youth (CBD decision XI/8 B paragraph 1).
- Involve civil society (CBD decisions XI/8 C paragraph 1) and Indigenous and local communities (CBD decisions XI/14 B paragraph 17).

Box 6 11 Coordinating action against invasive alien species in Great Britain: Great Britain Invasive Species Strategy and integration with plant and animal health.

The Great Britain Invasive Non-native Species Strategy (Secretariat, Great Britain Non-native Species, 2015) is an example of a National Invasive Alien Species Action Plan. It sets the strategic vision and national objectives for invasive alien species management in Britain and identifies 59 key actions to achieve their delivery. It is a partnership document developed by a combination of government bodies, environmental non-governmental organizations and organizations representing trade and industry.

The implementation of the Great Britain Strategy is overseen by a Programme Board comprising eleven government departments and delivery bodies. This reflects the wide range of threats posed by invasive alien species. The work of this Board is facilitated by a small secretariat (3.6 staff), which acts as a point of contact for stakeholders, establishes working groups on behalf of the Board to deliver specific actions, coordinates communications activity and runs a risk analysis mechanism.

The strategy has been broadly successful and has led, among other things, to the development of an invasive alien species risk analysis mechanism to support the ban on sale of invasive alien species; action plans to tackle key pathways of introduction; two awareness raising campaigns to reduce spread of aquatic organisms and ornamental plants; contingency plans and rapid responses; and a network of local action groups established to help tackle more widespread species (**Glossary**) in their local area. Examples of success include the eradication of five species from Britain (*Vespa mandarinia* (northern giant hornet), *Xenopus laevis* (African clawed frog), *Pimephales promelas* (fathead minnow), *Ameiurus melas* (black bullhead), *Lithobates catesbeianus* (American bullfrog)) and the ongoing eradication campaigns for three others (*Ludwigia grandiflora* (water primrose), *Myiopsitta monachus* (monk parakeet) and *Pseudorasbora parva* (topmouth gudgeon)). Britain has pioneered one of the largest invasive alien bird eradications in the world. In order to protect the indigenous *Oxyura leucocephala* (white-headed duck), the alien *Oxyura jamaicensis* (ruddy duck) has been reduced from a peak of over 6,000 individuals in 2001 to a handful of individuals in 2015 (Handerson, 2009; Secretariat, Great Britain Non-native Species, 2015).

However, despite these notable successes, overall indicators show the strategy is having little impact on the total numbers of invasive alien species establishing and spreading in Britain. There are about 2,000 alien species established in the United Kingdom, 10-15 per cent of which are invasive. Despite the Great Britain Strategy this number is increasing by

Box 6 11

approximately 10-12 new species per annum. An independent parliamentary enquiry (UK Parliament, 2019) attributed this increase to the dearth of resources available to tackle invasive alien species, including the lack of an inspectorate to prevent and intercept incursions. Only 0.4 per cent of Great Britain's total biosecurity budget is spent on invasive alien species despite their being similar, in terms of numbers of harmful organisms, to animal and plant health regimes. Unsurprisingly, other biosecurity regimes with much greater funds and dedicated inspectorates have been largely successful in their objectives by comparison. Indeed, animal and plant health regimes in Britain have prevented the introduction of 98 per cent of listed species in the past 20 years. By comparison, with no invasive alien species inspectorate, attempts to stem the flow of invasive alien species into Britain have been largely unsuccessful, with approximately 25 new invasive alien species establishing in the last 20 years.

Since 2013 the Department for Environment, Food and Rural Affairs in the United Kingdom has been working to develop a more integrated approach to biosecurity, incorporating animal health, plant health and invasive alien species. This has included establishing monthly meetings to review new and developing biosecurity threats across these regimes. Meetings are chaired by the Minister for Biosecurity, Marine and Rural Affairs, and attended by the Chief Vet, Chief Plant Health Officer and Chief Non-native (Alien) Species Officer. To support these meetings, emerging threats from invasive alien species, pests and diseases are reviewed within the same risk matrix. The matrix uses information from existing risk assessments to place organisms according to likelihood of an outbreak and potential impact. Impact is assessed using standardized criteria for economic, environmental and human health, which are then monetized to produce a single metric. This approach provides a straightforward overview of changing biosecurity threats and allows the minister and officials to compare and prioritize threats for action. It has resulted in greater integration across biosecurity regimes and the opportunity to utilize the greater experience of plant and animal health teams to support response to invasive alien species.

Strong and implementable NBSAPs, aligned with international regulatory frameworks, can help to spur the strategic actions and establish the properties required for the successful prevention and control of biological invasions, in alignment with the Kunming-Montreal Global Biodiversity Framework (CBD, 2022a) invasive alien species target (Target 6). Furthermore, coordinated efforts to strengthen national regulatory instruments can help address for online trading (aligned with Target 5), the creation of appropriate policies for the development and use of responsible environmentally sound technologies (aligned with Target 17), as well as making available data and information accessible (aligned with Target 21).

It is important to highlight that national action plans should implement existing international standards as a minimum standard but could also take full advantage of the rights under international agreements that allow for stricter protection measures. Also, a key instrument of national action plans is the possibility of creating departments or agencies specifically dedicated to the governance and implementation of invasive alien species legislation. By taking these steps, national strategies can define instruments and processes to ensure the need for shared efforts and commitments, and the understanding of the specific roles across sectors and Indigenous Peoples and local communities and multi-scale coordination of response programmes.

6.3.3.2 Careful delineation of legal authorities that would enable the implementation of risk assessment and surveillance protocols

The nature of the risks of invasive alien species depends on the stage of the biological invasion (Epanchin-Niell, 2017; Springborn *et al.,* 2011). In the introduction stage, preventing or minimizing alien species arrivals is achieved through actions that address both intentional and unintentional introductions. Prevention is also best achieved when clearly defining "who" has the authority for the detection of potential invasive alien species and understanding which responsible bodies of the legal and regulatory framework, such as Phytosanitary (defined under the IPPC) and animal health (defined under WOAH) mandates, allow actions. Actions designed to prevent establishment of potential invasive alien species are termed "early detection and rapid response" (EDRR), with the desired response eradication of the incipient invasive alien species (Meyerson & Simberloff, 2020).

Policies aimed at preventing or minimizing the possible effects of intentional introductions need to consider who is the responsible body for providing transport and introduction permits. The actions of these offices should focus on providing permits for proposed planned introductions and should be based on information coming from risk assessment procedures such as the Australian Weed Risk Assessment (Pheloung *et al.,* 1999), the Plant Protection and Quarantine weed risk assessment (PPQ WRA; Koop *et al.,* 2012) and the Non-native Species

Secretariat (NNSS) Risk Assessment Scheme for Great Britain (NAPRA Network, 2010). In the context of transport and introduction, a policy of inspection and/or treatment at the port of departure can duplicate the effect of other policy of inspection and/or treatment at a port of entry. However, if a departure and arrival policy is established, policy integration would require an international agreement. A key part of such agreement is aligning the authorities with the mandate of inspection and/or treatment.

6.3.3.3 Embed both surveillance and monitoring into policy instruments focused on invasion management

Invasive alien species can cross borders and, therefore, preventing their introduction can only be achieved with pre-border, border and post-border surveillance systems (Anderson *et al.,* 2017; Poland & Rassati, 2019). Successfully controlling invasive alien species or preventing biological reinvasions relies on long-term monitoring for early detection and rapid response (Amorim *et al.,* 2014; European Environment Agency, 2010b; Franklin *et al.,* 2011; Oswalt *et al.,* 2021; Roy & Roy, 2008). These activities are essential to minimizing their impacts, developing economically efficient and ecologically relevant management programmes and promoting citizen engagement and educational outreach (McGeoch & Squires, 2015; Oswalt *et al.,* 2021). Furthermore, these activities are at the core of CBD Guiding Principle 5: Research and monitoring (CBD, 2002) and are considered a fundamental tool to address the problem of invasive alien species. Continuous monitoring systems that use essential biodiversity variables (EBVs; **Glossary**; **section 6.6**) can also help evaluate the effectiveness of policy and management strategies and fill a fundamental knowledge gap in environmental policy.

For example, the European Union Regulation 1143/2014 on invasive alien species (Council of the European Communities, 2014) has specific provisions for member states regarding the implementation of surveillance systems to detect the presence of alien species of Union concern as early as possible and take rapid eradication measures to prevent their establishment. However, as noted by Latombe *et al.* (2017), in 2010, only 26 per cent of countries reported the establishment of national surveillance systems and monitoring activity. Additionally, the capacity to detect and react promptly to new invasions or re-invasions is often limited (see, for example, Genovesi, 2005) and is usually not comparable across countries (Latombe *et al.,* 2017). Furthermore, when monitoring takes place, it occurs at multiple unconnected scales – from national programmes to local citizen science (**Glossary**) initiatives (McGeoch & Squires, 2015; Oswalt *et al.,* 2021), which further complicates their interoperability. As a result, which agency oversees surveillance and monitoring activities, and where/when/how such activities occur, varies widely between countries.

Technical scientific bodies must be tasked with continuous monitoring activities, diagnosis, risk assessment, storage and circulation of information, reporting, identification and enforcement of appropriate responses. At a higher level, a global monitoring system is a critical tool to be included in multilateral environmental agreements to manage biological invasions effectively. This goal is within reach, as pointed out by Latombe *et al.* (2017).

6.3.3.4 Develop policy and regulatory instruments to underpin innovative management programmes

New technologies can be developed or translated from other contexts to improve any aspect of innovative management programmes for biological invasions (van Rees *et al.,* 2022). Innovation is the translation of invention through proof-of-concept to readiness to be deployed, leading to desirable outcomes (Baregheh *et al.,* 2009). Frameworks can drive innovation in management of biological invasions, particularly in the context of public good outcomes (van Rees *et al.,* 2022). Indeed, government support is also needed to find better solutions to management challenges from idea and blueprint to a full technology readiness level. Not all the technologies needed are available, nor is it clear what future valuable technologies might be. Only through policy development will governments invest in technology development and deployment (Burke *et al.,* 2005).

Innovation can be achieved with cultural change through community acceptance of the value of the technology interacting with institutional change, aligning to regulate technology deployment (Stilgoe *et al.,* 2020). Achieving cultural acceptance of technology is not guaranteed and often hinges on obtaining social license and acceptability and demonstrating that the benefits of the technology outweigh any risks, including ethical considerations. Cultural acceptance is never a fixed position (Crowley *et al.,* 2017b). Institutional change includes the necessary policy or regulatory environment that will regulate use (Burke *et al.,* 2005). An example was the invention and adoption of chemical pesticides for pests, weeds and diseases from the 1940's. Starting with dichlorodiphenyltrichloroethane (DDT), the consequences of its initial use led to completely national risk-based legislation and regulation on how when and where it could be used. This eventually resulted in DDT being banned as evidence of negative impacts became available (Mansouri *et al.,* 2017). This nonetheless opened the door for development and application of future generations of less toxic chemicals to which the initial regulations needed to be adapted (Handford *et al.,* 2015). Even the most benign chemical pesticides have now increasingly lost public favours based on long-term evidence mediated through changing cultural acceptance (Kudsk & Mathiassen, 2020). The same processes are necessary in the adoption of any new technology however beneficial, so it is the role of government to manage and respond to this culturally driven policy and regulatory process, without which any benefits from new technologies towards management of biological invasions will not be obtained.

6.4 ENGAGEMENT AND COLLABORATION WITH STAKEHOLDERS AND INDIGENOUS PEOPLES AND LOCAL COMMUNITIES

The engagement of stakeholders (**Chapter 1**, **section 1.5.1** for the definitions of stakeholder groups) and Indigenous Peoples and local communities can help construct coherent policy and management plans that are appropriate to local environmental and cultural realities (e.g., Adriaens *et al.,* 2015; Bravo-Vargas *et al.,* 2019; Bryce *et al.,* 2011; Dehnen-Schmutz *et al.,* 2010; Fischer *et al.,* 2014; Gaertner *et al.,* 2017; García-Llorente *et al.,* 2008; S. Liu *et al.,* 2010; Marchante *et al.,* 2017; Novoa *et al.,* 2016; M. S. Reed, 2008; M. S. Reed *et al.,* 2009; M. S. Reed & Curzon, 2015; Shackleton, Adriaens, *et al.,* 2019; Stokes *et al.,* 2006; Touza *et al.,* 2014). As outlined earlier (**Chapter 5**, **section 5.6.2.1**), there are many examples where effective regulatory, social responsibility and incentive-based systems have and continue to support effective industry and landowner engagement in the prevention and control of invasive alien species. This section outlines the context and summarizes the purposes of engaging stakeholders and Indigenous Peoples and local communities (**section 6.4.1**). It assesses the general enabling factors for successful engagement (**section 6.4.2**), as well as factors contributing to successful collaboration with Indigenous Peoples and local communities on biological invasions more specifically (**section 6.4.3**). Finally, the section explores the different governance network options for collaborative action (**section 6.4.4**).

6.4.1 Reasons for inclusive engagement

New approaches to governance for biological invasions reflect broader shifts in environmental governance (Chaffin *et al.,* 2014), emphasizing the interconnectivity of ecological and social systems and the uncertainty associated with complexity and rapid environmental change (**section 6.7**). These approaches recognize the need for more integrated multi-level or "polycentric" governance (Anderies *et al.,* 2013; Bodin, 2017; Lubell, 2013; Ostrom, 2010) and democratic legitimacy (Stoett *et al.,* 2019), as well as the imperative to achieve societal consensus, engagement and fairness (CBD, 2020a). The general shift is toward greater consideration of the adaptive-collaborative governance model (**section 6.2.4.1**), where a collective decision-making process is one "that allows diverse sets of actors who share an interest or stake in a policy or management issue to work together toward mutually beneficial outcomes" (Lynch, 2020).

As an early example, **Box 6.12** describes how New Zealand's biosecurity legislation underpins new collaborative forms of governance of invasion pathways in marine areas.

There has been no comprehensive review of the on-the-ground experiences of stakeholders and Indigenous Peoples and local communities or their engagement in management and governance of biological invasions (but see Shackleton *et al.,* 2019 for a review of stakeholder involvement in invasive alien species research). The following reasons for engagement are taken from a limited selection of the literature based on experiences with governance and management of biological invasions:

Knowledge-related

Engaging with stakeholders and Indigenous Peoples and local communities facilitates knowledge and information sharing, creating information flows across scales (Lansink *et al.,* 2018). This contributes to the development of sufficient shared awareness of biological invasions, including understandings of drivers, processes, impacts and possible responses (Carter *et al.,* 2021). Bridging different knowledge systems associated with different disciplines and perspectives (Barney *et al.,* 2019) provides greater legitimacy of knowledge underpinning actions and leads to higher quality and more context-relevant decisions (J. M. Evans *et al.,* 2008; M. S. Reed & Curzon, 2015). This also permits social learning for adaptive management that can assess and reduce uncertainty, build adaptive capacity (S. Liu & Cook, 2016; Maclean *et al.,* 2018; Novoa *et al.,* 2016; Söderström *et al.,* 2016), make monitoring more cost effective (Novoa *et al.,* 2016) and facilitate collaborative research (Shackleton, Adriaens, *et al.,* 2019).

Risk assessment

Engaging with stakeholders and Indigenous Peoples and local communities facilitates decision-making about risk prioritization by addressing people's concerns so that: (1) associated uncertainty can be discussed (S. Liu *et al.,* 2011), (2) diverse values and perceptions can be brought into the risk assessment/decision process to negotiate consensus, (3) risk can be contextualized in broader contexts, (4) measures targeted to the local context can be formulated, (4) decision-making becomes transparent and

Box 6.12 New Zealand's shift towards adaptive collaborative governance for biological invasions.

In part due to the recognition that reactive species-specific management designed for terrestrial invasions is not suited to the marine environment, New Zealand made a major shift towards adaptive collaborative governance. Legislatively enacted in a 2012 amendment to the Biosecurity Act 1993, governance for biological invasions was moved away from species-specific management to a proactive focus on vectors (**Glossary**) and invasion pathways that could better serve to prevent establishment rather than undertake costly remedial action. The first Marine Regional Pathway Management Plan, developed in Fiordland, a World Heritage Site of the United Nations Educational, Scientific and Cultural Organization (UNESCO; **Figure 6.14**), was "driven by a community-based, multi-stakeholder and Indigenous Peoples and local communities and government agency partnership" initiated by the Fiordland Marine Guardians, a stakeholder and Indigenous Peoples and local communities group composed of "representatives of Ōraka Aparima Rūnaka Inc of Ngāi Tahu iwi, commercial fishers, recreational fishers and charter boat operators". The "Guardians" are responsible for the integrated management of the Fiordland Marine Area and were officially empowered by national legislation to advise and recommend government and management agencies on all aspects of management, facilitate and promote integrated management, prepare and disseminate information and monitor and advise on threats, among others (Cunningham *et al.,* 2019).

Figure 6.14 **Fiordland National Park.**

The Marine Regional Pathway Management Plan developed in Fiordland is an example of adaptive-collaborative governance. Photo credit: Bernard Spragg. NZ from Christchurch, New Zealand – Milford Sound New Zealand., WM Commons – Public domain.

(5) complex trade-offs between different options can be assessed (Carter *et al.,* 2021; S. Liu & Cook, 2016; Moon *et al.,* 2015). For example, the Mohawks of Kahnawá:ke (Quebec, Canada) oppose spraying chemicals on the land as they feel it would contradict their spiritual connection with nature (IPBES, 2022). There can also be concerns over animal welfare and rights when considering possible management and eradication interventions **(Box 6.13)**. A particular trade-off to consider is the need for rapid response to maximize the likelihood of eradication early in the invasion curve (**Glossary**), *versus* the time and cost of broad stakeholder inclusion in decision-making. This trade-off can however be managed by ensuring as far as possible that collaborative and inclusive decision-making structures are in place before new invasive alien species arrive.

Consensus building

By increasing their involvement, developing shared aims, overcoming barriers to coordination and collaboration, and building trust (Carter *et al.,* 2021; Graham, 2019; Lynch, 2020), engagement with stakeholders and Indigenous Peoples and local communities can reduce and help manage conflict (Moon *et al.,* 2015; Shackleton, Adriaens, *et al.,* 2019), including around "conflict species" (Woodford *et al.,* 2016; **Chapter 5**, **section 5.6.1.2**). It increases public support, acceptance, ownership and buy-in (Lansink *et al.,* 2018), which minimizes the risk of unintended consequences and avoids the costs of failed measures (S. Liu & Cook, 2016; M. S. Reed & Curzon, 2015). Engagement can help to better manage the unequal distribution of costs and benefits (Novoa *et al.,* 2016) by appropriately sharing and differentiating responsibilities for management and increasing enforcement capacity (S. Liu & Cook, 2016; **section 6.7**).

Economic effectiveness and efficiency

Engaging with stakeholders and Indigenous Peoples and local communities ensures that interventions are efficient, equitable, and provide the correct incentives (e.g., that are not perverse for some or over-reward other stakeholders). This also increases the participation of economic stakeholders such as land owners, small businesses and corporations, and can help avoid the "tragedy of the commons" (**Glossary**; McAllister *et al.,* 2015).

Decision-making under uncertainty and complexity

Engaging with stakeholders and Indigenous Peoples and local communities enables adequate characterization and management of complex problems. It can also help to find solutions when conflicting perspectives, objectives and management goals make invasion problems difficult to characterize or resolve (J. M. Evans *et al.,* 2008; Woodford *et al.,* 2016). It allows a balancing of social, economic and ecological sustainability objectives and values across multiple interrelated interests (Carter *et al.,* 2021), leading to better decision-making under a diversity of local contexts. Action can then be adapted to local ecological, social and political contexts (McAllister *et al.,* 2015).

Coordination

Engaging with stakeholders and Indigenous Peoples and local communities facilitates coordination in complex situations between many apparently independent groups that have a wide range of interests, motivations and resources and who are directly or indirectly engaged in some aspect of invasions at multiple spatial scales (Barney *et al.,* 2019; Dandy *et al.,* 2017). It may help to create "institutional fit" within scales, bridge scales and develop cross-scale interactions to match the multiple scales of the problem (McAllister *et al.,* 2015). It can also help to meet multiple goals of different stakeholders with different needs for nature's contributions to people arising from the same ecosystem (Failing *et al.,* 2013; D. M. Martin *et al.,* 2018; Nel *et al.,* 2016). Responses to biological invasions may indeed need or benefit from the combination of public and private assets and joint actions by public and private sectors. Better coordination avoids competition and free-riders and contributes to reducing overall costs (Failing *et al.,* 2013; D. M. Martin *et al.,* 2018; Nel *et al.,* 2016), or shifting costs of management or impacts from one community, stakeholder to another.

Respect for rights, fairness

Engaging with stakeholders and Indigenous Peoples and local communities promotes democratic governance, where stakeholders have a direct voice in decisions that affect their environments and lives. It also encourages public engagement in publicly-funded efforts, influences decisions that affect people's good quality of life, and ensures accountability and fairness (Carter *et al.,* 2021; J. M. Evans *et al.,* 2008; S. Liu & Cook, 2016; M. S. Reed & Curzon, 2015). Finally, it ensures compliance with governance directives and stakeholder and Indigenous Peoples and local communities demands and rights for engagement (IPBES, 2022).

Box 6 13 **Invasive alien species control and animal rights.**

When invasive alien species are prioritized for control, ecological and economic aspects often take precedence, whereas species welfare can be underappreciated. In cases where the invasive species is associated with human values, and the control mechanism is lethal, unaddressed ethical issues in management actions can create conflict between stakeholders and Indigenous Peoples and local communities, including animal welfare groups and invasive alien species managers (**Figure 6.15**; **Chapter 5**, **Box 5.13**). Such conflict has been observed during lethal control of invasive *Erinaceus europaeus* (European hedgehog) on the Scottish island of South Uist (Warwick, 2012), of *Equus caballus* (horse) in parts of Northern America (Bhattacharyya & Larson, 2014), of *Trichosurus vulpecula* (brushtail possum) in New Zealand (Beausoleil *et al.,* 2016), and of the introduced *Epiphyas postvittana* (light brown apple moth) in parts of the United States of America (Zalom *et al.,* 2013). Such conflict delays invasive alien species control programmes, thereby potentially escalating the impact on native biodiversity, a point which is often not acknowledged by the parties in conflict (Russell *et al.,* 2016).

Consensus amongst invasive alien species managers, stakeholders and Indigenous Peoples and local communities, including animal welfare groups, has often been achieved through informed conversations. For example, in parts of the United States managers involved animal welfare groups early in the process of management to gain support for lethal control of invasive alien *Sus scrofa* (feral pig; Perry & Perry, 2008). Furthermore, involving local people while planning invasive alien species control measures can not only spread awareness

Box 6 13

about the severe impacts of such species, but also result in socially acceptable mechanisms for controlling invasive alien species. For example, government and indigenous community co-management of Kakadu National Park in Australia resulted in acceptable control of invasive feral pigs (Robinson *et al.,* 2005). These examples suggest that management of biological invasions can be achieved with horizontal integration of different sectors and vertical integration of different governance scales (**section 6.3**). Latent development of public awareness on invasive alien species control and use of popular media to communicate evidence (or curb misinformation) during a conflict-like scenario remains central to inclusive and successful invasive alien species management programmes (Crowley *et al.,* 2017a).

Figure 6 15 **Public display to support animal rights.**

Public display of a message by People for the Ethical Treatment of Animals (PETA) in Australia about responsible ownership of pet cats to stop them turning feral and getting killed in a lethal control programme. Photo credit: PETA Australia – under license CC BY 4.0.

6.4.2 Options for improving engagement with invasive alien species-related activities

6.4.2.1 How Indigenous Peoples and local communities participation can be better integrated with national policies and global efforts

Collaboration between Indigenous Peoples and local communities and other stakeholders is an underlying theme in calls for their participation in invasive alien species management efforts (S. M. Alexander *et al.,* 2017; Peltzer *et al.,* 2019; Reo *et al.,* 2017). This collaboration could take the form of "a strong and sustainable institution that can raise awareness, mobilize communities and design appropriate management plans" (**section 6.7**; Tilahun *et al.,* 2017), including:

- Use of community-led institutions that can bring together community members and make rapid decisions to respond to change, manage or pool communal resources, build leadership, facilitate interaction and demonstrate practices (Guneratne, 2002);
- Collaboration between Indigenous Peoples and local communities, researchers and government officials to prevent, detect and respond to invasive alien species, and
- Efforts to ensure the conservation of habitats on indigenous lands and leadership in biodiversity conservation and facilitate voluntary partnerships (Schuster *et al.,* 2019). For example, an agreement was signed between Sami people and the Norwegian government, grounded in the International Labour Organization Convention on Indigenous and Tribal Peoples in independent countries. This provides Sami

people with the right to be consulted on all matters of importance for the Sami (Broderstad & Eythórsson, 2014), using integrated approaches in which potential methods of control are implemented on a case-by-case basis (Broderstad & Eythórsson, 2014).

6.4.2.2 Factors contributing to failure and success of engagement with stakeholders and Indigenous Peoples and local communities

There are now many documented examples of stakeholders and Indigenous Peoples and local communities engagement processes that have failed to deliver intended outcomes for the environment, or even led to negative unintended consequences (Coglianese, 1997; Cooke & Kothari, 2001; Gerrits & Edelenbos, 2004; Lane & Corbett, 2005; Staddon *et al.,* 2015), for example, by inflaming latent conflicts (Emery *et al.,* 2015; Redpath *et al.,* 2013). As a result, criticisms of engagement practice have ranged from tokenism, where participants are manipulated to legitimize decisions, to broader critiques that key groups may not have the information, skills or equality needed to participate in effective governance, knowledge sharing, or learning processes (**section 6.4.4**).

There is no reliable "one-size-fits-all" blueprint for collaborative governance (**section 6.2.3.1**), but in response to these criticisms, Reed *et al.* (2018) distinguished different levels of engagement, and factors that might in theory explain why engagement does or does not deliver intended outcomes. Levels of engagement can be adapted to the purpose and context of the process, rather than necessarily aiming for the highest levels of engagement, such as co-production (**Figure 6.16)**. A conceptual model of participation that is inclusive and empowering can be used to build stronger participation and engagement (Bell & Reed, 2022):

- Before: Consider the role of factors that precede an effective participatory process (e.g., the creation of safe spaces and overcoming barriers to engagement to ensure the process is inclusive, including women and other marginalized groups).
- During: Take into account the factors that affect empowerment during the engagement process (e.g., equality between participants that respects and values different knowledge systems – including local knowledge and experience of invasive alien species) and agency, including freedom (from fear) and access to the resources and other means necessary to actively participate.
- After: Foresee factors that may continue to build empowerment or disempower participants after the process has concluded (e.g., accountability, ensuring decisions are implemented and reflect outcomes from the process and feedback loops that inform people how

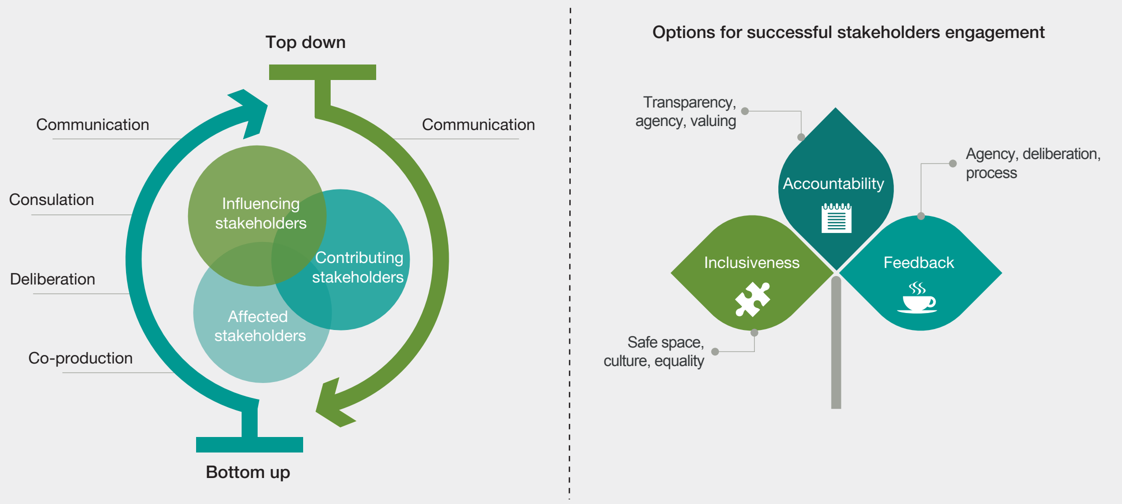

Figure 6 16 **Options for successful engagement of stakeholders and Indigenous Peoples and local communities.**

This figure shows the different options available to engage with stakeholders and Indigenous Peoples and local communities (left), and a focus on empowerment (right). Adapted from Reed *et al.* (2018), https://doi.org/10.1111/rec.12541, under copyright 2017 Society for Ecological Restoration and Bell and Reed (2022), https://doi.org/10.1093/cdj/bsab018, under license CC BY 4.0.

their knowledge has been used to manage invasive alien species in their area or sector; **Figure 6.16**).

An understanding of stakeholders and Indigenous Peoples and local communities' influences and interests, how each are likely to be involved in different invasion stages, and their participation as equal partners, can therefore contribute to the success of any attempt to represent, empower people and co-design biological invasion management and governance. If communicative and consultative approaches can deliver significant benefits (Shackleton, Adriaens, *et al.,* 2019), co-productive approaches may be more appropriate than hierarchical governance approaches, especially in contexts where there is significant conflict of interest or mistrust between stakeholders.

6.4.3 Coordination, collaboration and Indigenous Peoples and local communities[11]

Indigenous Peoples and local communities often have detailed knowledge of invasive alien species (**Chapter 1**, **section 1.6.7.1**), including their dynamics (**Chapter 2**, **Box 2.6**), drivers (**Chapter 3**, **Box 3.14**), impacts (**Chapter 4**, **section 4.6**), and the ability to manage or adapt to their presence (**Chapter 5**). Many Indigenous Peoples and local communities also have their own customary governance systems and institutions that may already be working to support management of biological invasions. There have indeed been many cases where management plans or techniques have negatively impacted the food security, culture and values of Indigenous Peoples and local communities (IPBES, 2022). This section will discuss efforts to avoid this scenario.

6.4.3.1 Rights of Indigenous Peoples and local communities

In some contexts, there are legal obligations to recognize the rights of Indigenous Peoples and local communities to manage their own lands and waters. The adoption of the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP) in 2007 has provided a specific framework for engaging with Indigenous Peoples. Free, prior and informed consent (FPIC) is a specific right that pertains to Indigenous Peoples, which allows them to give or withhold consent for a project that may affect them or their territories, which could include efforts to manage or eradicate invasive alien species. Furthermore, free prior and informed consent enables them to negotiate the conditions under which a project will be designed, implemented, monitored and evaluated. The framework is less clear for "local communities", but some countries have specific legal frameworks for working with specific groups.

Many Indigenous Peoples and local communities have requested that their own customary governance systems and institutions be recognized within efforts to manage biological invasions, recognizing that these systems can be strengthened and, in some cases, revitalized by support from outside institutions (IPBES, 2022). Recognition and clarification of land tenure, including access and ownership of land, waters and biological resources can support Indigenous Peoples and local communities to manage biological invasions (IPBES, 2022; Kamelamela *et al.,* 2022).

These specific knowledge and governance systems and normative frameworks often mean that Indigenous Peoples and local communities do not wish to be considered or approached in the same ways as other "stakeholders" discussed above, as such broad multi-stakeholder processes can often serve to diminish their participation and obscure their rights and goals. A deeper engagement with their knowledge and customary governance systems within rights-based frameworks and in accordance with national legislation can therefore benefit both communities' good quality of life and biological invasions management strategies.

6.4.3.2 Co-production of Indigenous Peoples and local communities planning and Biocultural community protocols

Despite the reasons for engaging with Indigenous Peoples and local communities and their knowledge and governance systems discussed above, many discussions of Indigenous Peoples and local communities rely on a "vulnerability narrative" which considers them as passive victims of damage from environmental change. This can lead to policies and management actions that interfere with community wellbeing and do not support capacity-building and cultural continuity (Reo *et al.,* 2017; **Chapter 4**, **Box 4.14**).

Many cases (70 per cent of reviewed cases)[11] suggest that, even where collaborations between outsiders and Indigenous Peoples and local communities are reported to be successful, they do not necessarily consider Indigenous and local knowledge and governance. Some cases report that outsiders tend to focus instead on teaching Indigenous Peoples and local communities about management of invasive alien species using scientific methods. This can cause the loss of knowledge and important cultural practices of Indigenous Peoples (Sillitoe, 1998), as well as undermine long-term management success. There are however positive examples of the inclusion of Indigenous and local knowledge, community governance and institutions in the management of biological invasions.[11]

11. Data management report available at: https://doi.org/10.5281/zenodo.5760266

For example, forest scientists partnered with Indigenous Peoples and local communities in Michigan, United States to co-design invasive alien species control experiments using traditional ecological knowledge (Poland *et al.,* 2017). Indigenous Peoples and local communities were involved in decision-making processes for weed control in Western Australia. Rangers consulted Indigenous Peoples and local communities' elders about their work eradicating weeds and used "place centred" methods (Bach *et al.,* 2019).

Overall, key aspects that Indigenous Peoples and local communities have highlighted in relation to successful co-production and co-management include respect for community knowledge, institutions and protocols, allowing enough time to build trusting relationships, and broad distribution of benefits from biological invasion management, which do not need to be financial and can include capacity-building in research and management.[11]

Some Indigenous and local communities have developed biocultural community protocols (**Glossary**), documents that consider their values, procedures and priorities to frame how they wish to be engaged in projects that impact them. They set out rights and responsibilities under customary, state and international law as the basis for engaging with other stakeholders (Natural Justice, 2022). Biocultural community protocols could be a foundation for discussions with communities on policies related to managing invasive alien species and restoring ecosystems. For instance, in Hawaii, a biocultural community protocol has been developed to support the successful ecosystem restoration of the Puʻuwaʻawaʻa Community-Based Subsistence Forest Area (Kamelamela *et al.,* 2022). Co-production of planning and decision-making, or support of existing Indigenous Peoples and local communities' invasive alien species management systems could indeed benefit communities beyond biological invasions management. It provides recognition of their knowledge systems and incentives to continue or revitalize traditional monitoring, management and knowledge transmission and simultaneously enhances the efficacy of biological invasions management (IPBES, 2021, 2022)

6.4.4 Governance networks for collective action

Engaging with stakeholders and Indigenous Peoples and local communities, and considering diverse actors involved in governance for biological invasions (**section 6.2.3.3**) can be achieved by establishing informal or formal mechanisms for collective action (**Glossary**). Simply recognizing this need does not mean that collective action will happen, nor that collective initiatives and arrangements will be effective at solving the problems at hand (Koontz & Thomas, 2006; Lubell, 2004). Collective action outcomes often emerge through self-organization (where overall organization arises spontaneously from local interactions) – as when numerous individual managers acting independently apply cultural controls that together change the invasibility of landscapes (P. L. Howard, 2019). As demonstrated in an Australian rural landscape through a Landcare program, successful collective action emerged through the key role of a leader, building trust and social norms in the community, along with contracts that strengthened commitment and steered action towards the control of high priority invasive alien species (McKiernan, 2018).

Collective action is also often jointly planned and executed, based on place-based or culturally-based rules and norms where community members jointly assume responsibility for invasive species management (e.g., Graham *et al.,* 2019; Lien *et al.,* 2021; Lubeck *et al.,* 2019; Sullivan *et al.,* 2017; Yung *et al.,* 2015). However, the conditions required to engage in collective action are often absent; for example, an awareness of cross-boundary relationships, beliefs and expectations that other people will carry out appropriate actions, an absence of effective leadership or low confidence that collective efforts will be effective (Bodin *et al.,* 2019; Lubeck *et al.,* 2019; **Figure 6.24** in **section 6.7.3**). Even in cases where collective action mechanisms are in place and function well, there is often a need to support or coordinate such actions at higher levels of governance. Research has shown that a number of micro-interventions during community engagement can change opinions, beliefs and commitment leading to improved management outcomes (Niemiec *et al.,* 2019).

6.4.4.1 Coordination *versus* cooperation: challenges and options

It is important to differentiate biological invasion problems that are more related with coordination from those requiring cooperation (Bodin *et al.,* 2020). Effective coordination depends on finding ways for stakeholders who generally share the same viewpoints and interests to agree on how best to address a problem. This mainly involves mechanisms "to accomplish a generally agreed upon objective through, for example, efficient resource allocation, synchronization of different activities and a suitable division of labour for common tasks" (Bodin *et al.,* 2020). This is the case, for example, with *Fusarium* dieback (an invasive alien pathogen vectored by beetles that causes disease that damages avocado and more than 39 other tree species) in California, United States, which quickly prompted numerous government and non-government stakeholders who cooperate at different scales with very similar objectives to coordinate and confront the issue and develop a cohesive state-wide strategy (Lynch, 2020). **Box 6.14** presents another example of a governance network solution to achieve coordination between stakeholders who held similar interests in invasive

Box 6 14 Coordinating American mink eradication in North East Scotland through community partnerships and adaptive management.

In Northeast Scotland, when small-scale American mink removal projects failed due to mink recolonization from surrounding uncontrolled areas, adaptive-coordination strategies (**Table 6.1**; **section 6.2.3.1(4)**) were devised and implemented to eradicate mink over a large area. Due to uncertainties about the size of the mink population, mink's dispersal capacity, and the volunteer resources that would be needed and available to effectuate mink eradication, an adaptive management approach was developed involving formal coordination between diverse stakeholders. The project was initiated by scientists and supported by a government agency, a national park authority and local fisheries boards, all of whom shared an interest in mink eradication.

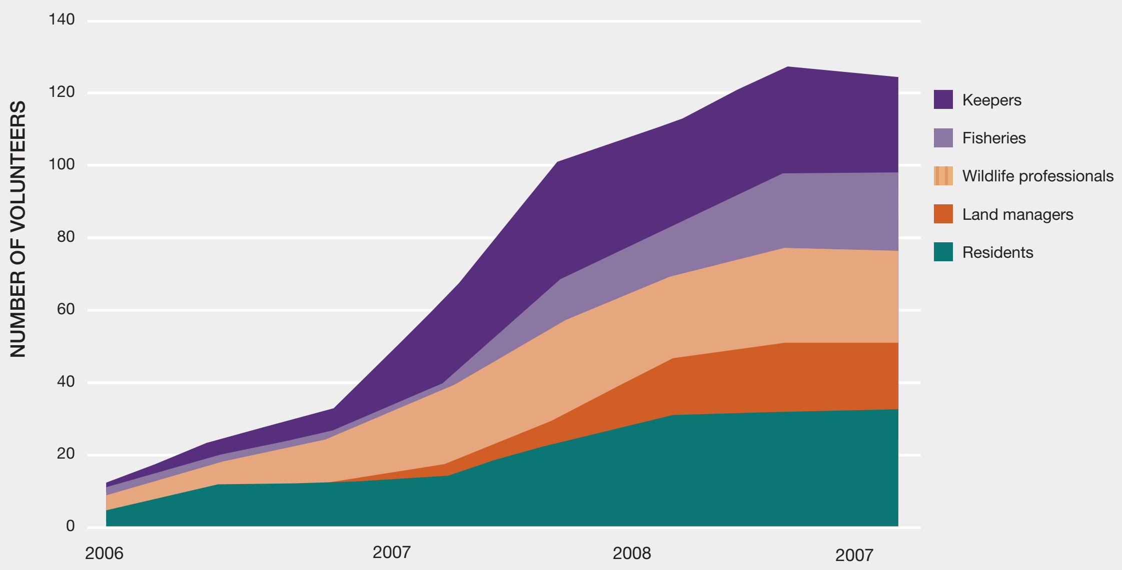

Figure 6 17 **Cumulative mink captures over the entire project area showing an increasing number of captures by volunteers, with the total number of active volunteers.**

Throughout the project shown by category. Source: Bryce *et al.* (2011), https://doi.org/10.1016/j.biocon.2010.10.013, under license CC BY 4.0.

Together, a coordinated coalition of trained volunteers was created to detect and trap mink **(Figure 6.17)**. These groups created a formal partnership that funded the project and provided in-kind contributions. The project achieved "multi-scale mink removal over 10570 km^2 with 10000 appearing to ¬be free of breeding mink 3 years after inception." Over time, the number of the local volunteers detecting and trapping mink grew – especially among local residents, land managers and wildlife professionals (Lambin *et al.,* 2012). "The defining factors underpinning the success of the project are strong volunteer involvement, efficient and systematic methods of monitoring and control, an adaptive approach to suit local conditions, the strategic use of topography to minimize recolonization and an ambitious vision; elements that are applicable to other invasive alien species and areas. It is a strong testament to what can be achieved when empowering local communities to take a stake in their local biodiversity and thus reason for optimism that the tide of invasion can be rolled back on a large scale where the convergent interest of local communities can be harnessed" (Bryce *et al.,* 2011).

alien vertebrate eradication, and where adaptive responses were needed to overcome uncertainties.

However, in many cases, invasive alien species present a cooperation problem rather than a coordination problem, as they involve stakeholders "with opposing interests seeking and finding agreeable ways to solve collective problems and dilemmas where their different interests are at stake, and where a solution often requires actors to make some sacrifices" (Bodin *et al.,* 2020). Thus, cooperation is associated with conflicts of interest, inherent trade-offs and subsequent tensions among actors. This was the case, for example, with the Panama Tropical Race 4 incursion in banana plantations in north Queensland, Australia. Many banana growers argued that their value- and cost-losses were not sufficiently compensated when implementing state-mandated biosecurity measures, leading ultimately to negotiations between the growers' industry body and the government on new standards and guidelines for production post-infection (Maclean *et al.,* 2018). Very often, there are multiple groups of "outcome winners" and "cost, value and collateral" losers whose interests and perceptions of invasive species are in conflict. Compared with coordination, cooperation is thus associated with higher risk for all of the actors involved (Berardo & Scholz, 2010). Thus, there is much greater need for processes of engagement that involve consensus-building, negotiation, knowledge integration and development of trust.

6.4.4.2 Tailoring collaborative governance networks

Collaborative governance networks consist of individuals and organizations that have come together to solve common problems that would be difficult or even impossible for a single organization to address alone (**section 6.2.3.3**). Governance itself can be characterized as a "polycentric network" of relations between government and non-governmental stakeholders whose knowledge, behaviour or interests are involved in different aspects of environmental policy-making (Berardo & Lubell, 2019; Bodin, 2017; Bodin *et al.,* 2017; Kluger *et al.,* 2020; Lubell, 2013). Such networks span a range of types, from "completely decentralized with all participants connected, to completely centralized with all collaboration brokered by a single organization" (Lubell *et al.,* 2017). Multiple types of networks across this span are likely to be needed for progress towards and implementation of integrated governance for biological invasions in each context (**section 6.2.3.1**).

The distinction between coordination and cooperation has strong implications for the type of governance network and collective action arrangements that are, more or less, suited to the tasks at hand. Such networks typically take three different forms, each with its own advantages and limitations (Provan & Kenis, 2008; **section 6.2.3.3**):

1. In participant-governed networks, there is no obvious central leading actor – all actors contribute fairly and equally to the collective effort. These networks tend to be dense (there are many social ties between the participants) and can work very effectively, but typically suffer when the number of members is high, since the lack of a hierarchical structure makes it difficult to coordinate numerous actors.

2. The second form is where one or a few of the participating actors take on a leading role, and the network subsequently takes a centralized "hub-and-spoke" structure where the leaders are directly connected to all or most other actors, thus becoming the hubs in a wheel-shaped network.

3. In the third form of network, a dedicated coordinating actor (a "Networker" stakeholder and a representative of Indigenous Peoples and local communities) is appointed as the central leader – a network administrative organization (NAO), also referred to as a bridging organization (Crona & Parker, 2012) or a collaborative institution (Lubell *et al.,* 2010). The network administrative organization can be created by the network members or provided or imposed externally. An external actor that wants to build a governance network to address invasive alien species could create a network administrative organization either to enhance and strengthen (and/or possibly control) existing governance networks, or to build a governance network from scratch.

An example of a successful network administrative organization (NAO)-led network management is found in the management of a hybrid between introduced *Sporobolus alterniflorus* (smooth cordgrass) and native *Sporobolus foliosus* (California cordgrass) in San Francisco Bay, a multi-tenure area where government agencies, private landowners and others are involved in efforts to eradicate this ecosystem engineer that will reinfest if it is not eliminated in all areas of the Bay. The management of *Sporobolus* is governed by a collaborative partnership between private landowners, government agencies and other stakeholders called the Invasive Spartina Project. The California State Coastal Conservancy, an agency whose mission is to protect the coast, and an environmental consultant founded the project with state funding. Together they serve as "central brokers" that coordinate the activities of stakeholders as a NAO. The project has "successfully removed 95% of invasive *Spartina [Sporobolus]*, and is now engaged in suppressing re-invasion and ecological restoration…the [project] has been very effective in comparison with other local collaborative partnerships" (Lubell *et al.,* 2017). However, such arrangements may not be effective for invasive alien species that have very different ecological dynamics, such as with very fast spreading marine species, where a myriad of organizations and individuals would be involved at short notice.

Hybrid networks may consist, for example, of bottom-up, participant-led initiatives that are provided with support and coordinated by a "networker" stakeholder. This is the case with the award-winning Victoria Rabbit Action Network (VRAN), which was developed in Victoria, Australia, in response to the failure of a largely top-down regulation and enforcement regime (L. B. Adams, 2014; L. B. Adams *et al.,* 2019; **Box 6.15**). In Central Burnett, Queensland, Australia, when the state reduced support for management of *Bactrocera tryoni* (Queensland fruit fly) and encouraged greater grower self-reliance, growers and their industry body formed an Area-Wide Management Committee, which acted as a network administrative

Box 6 15 **Case study illustrating a successful expansion and temporal shift in governance from top-down regulation to community-led action.**

A case that demonstrates the shift from top-down, regulation and enforcement based governance toward a government-supported, community based approach is found in a United Nations Award-winning initiative developed to manage one of Australia's most costly invasive vertebrates, the European rabbit (L. B. Adams, 2014). Prior to the development of the programme, information and power asymmetries limited the effectiveness of rabbit management, as those responsible for on-the-ground control – landowners and community organizations – were "kept at arm's length," at the same time that conflicting perspectives on the need to control rabbits, animal welfare concerns and changes in land use presented barriers to top-down regulation and targeted programmes (Reid *et al.,* 2019). The Victorian Rabbit Action Network (VRAN) was developed by the National Rabbit Facilitator collaborating with groups involved in rabbit management, and adopted "systems-thinking…to understand how rabbit management works from a range of perspectives, test assumptions, and…develop and test strategy ideas" (L. B. Adams, 2014). The initiative focused on building networks and improving information flows and knowledge sharing through knowledge brokers. Communities were seen as sources of "socio-political and technological innovation, as opposed to consumers" and innovation was stimulated through competitive grants (Reid *et al.,* 2019). A democratic, participatory approach not only allowed a diversity of perspectives and experiences to be shared – conflicts inherent in the process "served as a driver of innovation, in that differing perspectives were discussed in respectful and authentic ways, allowing the emergence of innovative ideas and new ways of working together." The evaluation by Allen (2017) of VRAN's activities found that communities were exercising their agency and acting collectively, with decreased reliance on government (Allen, 2017). In four years, over 5,300 people were surveyed and 84 per cent of respondents were using an integrated rabbit management approach. In part due to this success, an additional programme was introduced in 2017 to support coordinated community-led action around the use of a new strain of calcivirus (rabbit haemorrhagic disease virus, RHDV K5) for biocontrol (Reid *et al.,* 2019). Unfortunately, overall outcomes on rabbit numbers and impact were not monitored.

Checklist of principles and requirements for successful community-led action on rabbits (L. B. Adams, 2014):

1. Leadership with empowered community groups
2. A community owned vision, philosophy, purpose and narrative
3. A partnership approach among the institutions and groups involved in rabbit management, with joint decision-making, responsibility, action and resourcing
4. Coordinated planning and action guided by a strategy, with understanding of:
   a. Community concerns and motivations that are generating interest in rabbit management
   b. What the community seeks to achieve and can realistically achieve – short and longer-term
   c. Current management practices: adaptive natural resource management, integrated pest management (**Glossary**), consideration of longer-term options to reduce rabbit impacts which goes beyond a focus on reducing rabbit populations, consideration of regulation and compliance requirements, consideration of options to assess and monitor a rabbit problem and rabbit impacts, consideration of the short and longer-term benefits and costs of different control options
   d. The available resources and tools that can assist community planning, action, learning, awareness, education, leadership development and innovation
   e. How best to navigate institutional arrangements that affect community capacity and action on rabbits
   f. How best to focus resources and effort
5. Demonstration and celebration of results and success linked to the community's vision and purpose
6. Government support for compliance
7. Recognition that community-led action and collaborative strategies involving all groups with rabbit management responsibilities are critical for success

organization (NAO), together with government staff, local growers, the industry body and other local stakeholders. Among other activities, it carried out management trials, fine-tuned management to suit individual grower operations, provided resources for treatments in towns funded by voluntary grower contributions, transferred expert knowledge and engaged in awareness raising. Reported results were "spectacular" – peak trap catches prior to the new governance arrangements were 240 flies per trap each day, reducing to one fly within a few years (Kruger, 2016).

With respect to the relation between governance networks and cooperation or collaboration problems, it is argued that denser, overlapping networks reduce monitoring and sanctioning costs involved in resolving collective action problems, for example, if some members freeride (e.g., allowing other participants to do the bulk of the work) or do not cooperate. Coordination problems, on the other hand, favour more "open" network structures. However, "the advantage of central coordination declines with the complexity and need for consultation involved in crafting solutions" (Berardo & Scholz, 2010).

Participant-governed networks are more effective at addressing cooperation problems (albeit only for smaller networks), while centralized networks (with or without a network administrative organization) are better suited for coordination problems (Bodin, 2017). Recent research has nuanced the proposed relationship between network structures and coordination/cooperation problems by explicitly accounting for trust, costs and risks as intermediate factors (Bodin *et al.,* 2020). More research in varied biogeographic and socio-economic contexts will contribute to improve governance networks.

6.4.4.3 Collective action

Governance for biological invasions is in part a collective action problem with coordination or collaboration solutions (Hershdorfer *et al.,* 2007; Epanchin-Niell *et al.,* 2010; McLeod & Saunders, 2011; Bagavathiannan *et al.,* 2019) including, for example, public-private partnerships (Mato-Amboage *et al.,* 2019). As outlined earlier, the mobility of invasive alien species means that prevention of their movement and management of established populations can benefit from collaboration and coordination across property and jurisdictional boundaries (Graham, 2014; Yung *et al.,* 2015; T. M. Howard *et al.,* 2018). Achieving such cooperation is challenging because diverse actors have varying levels of interest, skills, resources, capacity and time to commit to management of biological invasions (Donaldson & Mudd, 2010; Graham, 2013; Ma *et al.,* 2018; Kropf *et al.,* 2020). Successful collective action can be achieved through stakeholder and Indigenous Peoples and local communities networks. Social norms and trust can be established to develop a common understanding of the problem, agree on a common goal, identify measures of success and encourage participation in individual and group activities (Stallman & James, 2015; Niemiec *et al.,* 2016; Graham & Rogers, 2017; T. M. Howard *et al.,* 2018; Bagavathiannan *et al.,* 2019). There are sets of useful questions that are consistent across these cooperative invasive alien species management initiatives **(Table 6.9)**. While consideration of these factors listed in **Table 6.9** does not guarantee success, the questions do provide practical insights into what collective action offers for improving the governance for biological invasions into the future.

Table 6 9 **Collective action questions towards improved governance for biological invasions.**

Leadership	**Who will lead the collective action initiative?**
Working relationships	Who to include?
	Are there existing relationships among participants?
	How can trust be built among group members?
Shared goal	What is the shared goal?
	What are the ecological, economic and social dimensions of the goal? (What motivates one person may not motivate another)
	Does the goal focus on a single species or a whole ecological community?
	What area does the goal cover?
	What does success look like?
Pooling resources	What are the asymmetries in the programme?
	Who needs support?
	How can such support be provided?
	Where will resources come from over the short and long term?
Coordination	How can disparate efforts be connected?
	Who will make the connections?
	What support can be provided across initiatives?
	How will gaps be filled?

6.4.4.4 Applications of network analysis to support adaptive-collaborative governance

Network analysis of governance structures and stakeholders is useful for understanding and improving cooperation, coordination and information flows across a large number of actors and organizations engaged in biosecurity. As an example, in Australia, there is a "recognition that government players are neither resourced sufficiently to fill all required roles and responsibilities, nor necessarily the most capable of filling all roles…biosecurity now turns to multiple purpose networks that seek to mesh diverse tasks such as surveillance, policy development, response to incursion, awareness building, and research and development" (McAllister *et al.,* 2020). An independent marine pest network was formed to provide continuous communication for surveillance and rapid response (i.e., a "suite of organizational interactions that emerge around the Government's Marine Pest Sectoral Committee"), and includes scientists, industry and members of the public. The network is focused on communication, surveillance and engagement but does not manage incursions. Social network analysis showed that the "network is well-structured for information dissemination and there is evidence that nongovernment actors already play some role in integrating and brokering information", however, improvement is required, as it was found that, while information is provided to the community, there is a "near absence of ties for receiving information from the community" (McAllister *et al.,* 2020).

Adaptive-collaborative governance thus involves multi-level and multi-actor coordination and collaboration based on knowledge and disciplinary integration, experimentation, monitoring, the use of the best available technology and social learning (Kirkfeldt, 2019; **section 6.2.3.1(4)**). Many frameworks have been developed that explicitly consider the interactions between social and ecological systems. Not all frameworks, however, give equal emphasis to humans and ecosystems or their dynamic interactions (Binder *et al.,* 2013). Those that do, insist that involving stakeholders in collaborative environmental governance of such systems is imperative. Some of these approaches focus on networks of human-ecological interdependencies either within bounded geographical areas or, more often, across geographical barriers or boundaries and governance realms. This includes, for example, marine protected networks that usually involve large geographic areas, ecological connectivity, and many different government agencies and stakeholder and Indigenous Peoples and local communities groups (S. M. Alexander *et al.,* 2017).

Network analysis may be used to examine both the "fit" between governance and ecological scales and processes and the need to achieve coordination and collaboration among a large number of government agencies and stakeholders. Multilevel horizontal and vertical governance ties bring actors together to form multilevel (local to national) networks to coordinate, collaborate and share knowledge that is needed, e.g., to address a lionfish invasion in Jamaica's marine protected areas **(Box 6.16)**. Another example of how collaborative governance networks can be established to achieve greater ecological and governance fit is presented by network administrative organization-led coordination of *Phragmites australis* (common reed) management in the Great Lakes, United States **(Box 6.16)**.

Box 6 16 **Two examples of network analysis and adaptive-collaborative governance to strengthen the prevention and control of invasive alien species.**

(1) Horizontal and vertical governance ties to achieve social–ecological fit in response to a lionfish invasion an emerging marine reserve network in Jamaica.

Invasive alien species context: *Pterois volitans* (red lionfish) and *Pterois miles* (lionfish) favour near-shore reef habitat and are now prevalent across the Caribbean, Gulf of Mexico and Western Atlantic. First sited in the Bahamas in 2004, populations and distributions expanded rapidly (Côté *et al.,* 2013). By 2009, their range had expanded to Jamaica, with populations becoming established around the entire island within a year (Schofield, 2009). With no natural predators, lionfish consume a significant amount of especially juvenile native fish, depleting near-shore fisheries and coastal biodiversity in Jamaica. A study in the Bahamas found that, over a 2-year period, an increase in lionfish biomass coincided with a 65 per cent decrease in the biomass of 42 native species (Green *et al.,* 2012).

Governance context: The Jamaican Government has established, to date, 17 "special fishery conservation areas" (i.e., marine no-take areas that range from about 1 km^2 to 18.73 km^2). The majority of these conservation areas are close to small coastal communities with active small-scale and artisanal fisheries that use mixed gear (e.g., fish traps, spear guns) and target multiple species (e.g., conch, lobster, reef fish). It also established co-management arrangements with local non-governmental organizations and/or fisher cooperatives that devolve roles and responsibilities (e.g., monitoring) associated with the day-to-day management of these marine reserves (S. M. Alexander *et al.,* 2015).

Box 6 16

A multi-actor network: Social network analysis revealed that the governance structure of the Special Fishery Conservation Areas is constituted of ties between actors, including government and non-governmental organizations (S. M. Alexander *et al.,* 2016). Many of the ties emerged through one of the following three processes: (i) formal partnerships (e.g., co-management arrangements, capacity-building); (ii) personal connections and relationships; and (iii) joint membership on committees, boards and projects. For example, the Lionfish Project – funded by the Global Environment Facility (GEF) – fostered network ties between government agencies, non-governmental organizations, community-based organizations and private resorts. Collectively, the resulting governance network provided a critical foundation for an island-wide lionfish monitoring and culling programme. Multi-actor network ties connect actors horizontally across local sites of action and management that are geographically distributed, which is essential when effective responses to a biological invasion occurs simultaneously across sites. However, research revealed a lack of strong ties and information sharing between those local level management organizations with a mandate to manage one or more special fishery conservation areas. Multi-actor governance networks that span sectors, departments and agencies can contribute to increased coordination, which is central for effectively responding to and governing biological invasions and the multiple dimensions of socioecological fit associated with marine protected areas. Multi-level network ties can be central to linking action at multiple scales and tightening feedback, which are critical processes for effectively responding to and managing biological invasions. Multilevel linkages played the greatest role in enhancing fit in the marine reserve network. However, the long-term propensity of the multi-actor and multilevel networks to enhance social–ecological fit is uncertain given the prevalence of weak social ties, lack of a culture of information sharing and collaboration and limited financial resources.

(2) An adaptive collaborative governance approach to *Phragmites australis* (common reed) management in the Great Lakes, United States.

An example of adaptive collaborative governance is presented by the Great Lakes Restoration Initiative,[12] which supports a basin-wide initiative called the Great Lakes *Phragmites* Collaborative (Braun *et al.,* 2016). *Phragmites australis* is an invasive alien wetland species that affects ecosystem functions, biodiversity and social and economic values, and threatens restoration, generating large financial burdens for land management. A total of $16 million was invested in the Great Lakes Restoration Initiative from 2010-2015 to support the Great Lakes *Phragmites* Collaborative, which was developed to address numerous barriers to *Phragmites australis* management, including a lack of organized communication among managers and between managers and researchers; a failure to address *Phragmites australis* at a landscape scale (multi-state and bi-regional), and a lack of a common agenda or strategic plan in a context where stakeholders working independently were producing isolated impacts or duplicating efforts, leaving gaps that undermined management.

As a “neutral facilitating” entity, the Great Lakes *Phragmites* Collaborative serves as a regional representative for impacted stakeholders, providing support to develop common agendas, mutually reinforce activities and share measurements for assessing progress. An advisory committee represents a diversity of disciplines and expertise across different state and non-state organizations and geographical areas. The committee articulated a vision statement and common agenda that will be elaborated on, as much as possible, by consensus, which also includes support for individual initiatives. An adaptive collaborative process “involves stakeholders progressing toward a goal through a structure that facilitates mutually reinforcing activities and regular feedback… aligned efforts, support for discovery of new practices, and widespread adaptation of successful practices…an adaptive management technique because it promotes learning and adaptation.” A shared measurement system is considered to be essential for adaptive management – to assess progress, align individual strategies with landscape-level goals, and provide empirical information for adaptation. It showcases best management practices and lessons learned, and responds to needs identified in a stakeholder and Indigenous Peoples and local communities survey by providing access to information resources, information sharing and technology transfer (Braun *et al.,* 2016).

Network analysis can also be applied to both the biophysical and social processes entailed in pathways of introduction or spread of invasive pathogens, plants, or mammals, which includes international and domestic transport and other types of movements involving stakeholders, agents (e.g., ships), events and species or hosts that interact in space and time (D. C. Cook *et al.,* 2010; Hulme *et al.,* 2018; Lansink *et al.,* 2018). Within-sector (e.g., livestock, forestry) network analysis is related to the movement of invasive vectors and hosts through chains that link vector stakeholders and Indigenous Peoples and local communities with contributors in supply or value chains (**Glossary**). The FAO promotes such analysis to provide an evidence-base for animal diseases epidemiology to inform risk analysis and develop strategic plans for disease control and surveillance (FAO, 2011). Contact networks (i.e., networks and linkages in value chains that connect production systems, markets and

12. https://www.greatlakesphragmites.net/

consumers) can favour the transmission of contagious diseases within and between sectors, and need to be taken into account in the development of risk management strategies for the control and prevention of animal diseases (FAO, 2011). The input of a wide range of stakeholders is essential for this network analysis to be effective.

6.4.4.5 Challenges of collaborative governance approaches and success factors

Governance approaches themselves can be a significant source of conflict in invasive species management, particularly when various groups of influencers or interested stakeholders and Indigenous Peoples and local communities are not consulted, their knowledge is not taken into account, or they are not involved in implementation actions that affect them (Crowley *et al.,* 2017a; Estévez *et al.,* 2015; Lynch, 2020). Adaptive-collaborative management benefits from good governance, and vice versa. Plummer *et al.* (2013) examined the literature on adaptive collaborative management for governance content and found multiple relationships: among others, good governance is necessary to facilitate adaptive collaborative management, which helps facilitate a shift to good governance and can operationalize governance, while stressing multi-level, multi-sector and multi-stakeholder and Indigenous Peoples and local communities engagement (**section 6.7**). Common themes that emerge include the need for: accountability and legitimacy, the involvement of diverse stakeholder groups and Indigenous Peoples and local communities and bridging organizations, the need to achieve organizational fit, interplay and scale; for adaptiveness, flexibility and learning, as well as social learning and knowledge sharing[13] (drawn together in **section 6.7**).

In addition to giving citizens and stakeholders a voice in decisions that affect them, it is claimed that collaborative approaches to environmental governance can reduce conflict, build trust and facilitate learning among citizens and stakeholders, increasing the likelihood that decisions are implemented on the ground and over the long-term (e.g., Beierle, 2002; De Vente *et al.,* 2016; Derak *et al.,* 2018; M. S. Reed, 2008; M. S. Reed *et al.,* 2018). However, stakeholders and Indigenous Peoples and local communities involvement can only be successful when tailored to the problem and context (**section 6.4.1**). In some cases, stakeholders and Indigenous Peoples and local communities need to be involved in "deeper, two-way, co-productive engagement (possibly over long time-scales)" (Shackleton, Adriaens, *et al.,* 2019). This may be the case, for example, when coordination of the management of biological invasions occurs across multiple land tenures or land-use settings (Bryce *et al.,* 2011; Shackleton *et al.,* 2015), or where cooperation problems are evident and thus the potential for conflict or lack of cooperation is high. These conditions may call for stakeholder and Indigenous Peoples and local communities' involvement in the co-design or co-development of risk assessments, strategies and management approaches, co-creation of knowledge and co-implementation.

Several factors seem to be key to the success of adaptive-collaborative governance for biological invasions. One of them is the breadth of involvement of stakeholders and Indigenous Peoples and local communities, ensuring that all stakeholders with influence and interests are included lends governance the legitimacy it needs for policy implementation. Another factor is the deliberative and transparent nature of the collaborative process, as well as its ability to account for and manage power imbalances and conflicts (Newig *et al.,* 2018). Finally, high levels of social interaction among the participating actors favour positive outcomes and help to build commitments, knowledge and trust (S. M. Alexander *et al.,* 2018; Bodin, 2017; Newig *et al.,* 2018). These in turn are instrumental for collectively addressing coordination and collaboration problems. In other words, one key factor that is needed to achieve successful collective action is to build appropriate governance networks where relevant actors, individuals and/or organizations are included and engaged with each other (DeFries & Nagendra, 2017). Other critical factors that affect the effectiveness of networks include consensus around goals and the need for "network competencies," or specializations, among the network's participants, such as research competence (Lubell *et al.,* 2017).

Engagement with stakeholders and Indigenous Peoples and local communities is therefore an essential element of integrated governance of biological invasions (**section 6.7**). While it may not be possible, due to time or resource constraints, to develop effective adaptive-collaborate governance networks, deep stakeholder and Indigenous Peoples and local communities engagement can be built into any governance and policy development approach.

13. Data management report available at: https://zenodo.org/doi/10.5281/zenodo.5762739

6.5 ECONOMIC AND FINANCIAL OPTIONS

Although the costs associated with invasive alien species have been estimated to be in the trillions of dollars globally (Diagne *et al.,* 2021; **Chapter 4**, **Box 4.13**), the economic, political and financial systems have not yet sufficiently internalized these estimates. Therefore, biological invasions remains largely unaddressed at the national and international level (Pimentel *et al.,* 2005). Many impacts are unrecorded due to serious data gaps in several regions and there are ongoing methodological challenges about how to estimate social costs. It is clear, however, that the costs of impacts far outweigh the costs of management (**Chapter 5**, **section 5.5.7**; Diagne *et al.,* 2021).

Of particular concern are the indirect impacts of invasive alien species, as they are both inherently difficult to quantify and, in some cases, magnified under the prism of climate change (Mainka & Howard, 2010). Invasive alien species pose an enormous risk to good quality of life through their effects on social, economic and environmental systems (**Chapter 4**, **sections 4.5** and **4.6.3**). In some cases, such as invasive alien species-related agricultural losses, these effects can potentially destabilize socioeconomic and democratic structures by causing famine and social unrest (Goss *et al.,* 2014; Singh & Kaur, 2002).

These indirect impacts are yet to be incorporated into national accounting measurements of economic growth (e.g., gross domestic product (GDP) and gross national product (GNP)). For example, economic growth measures include exports as a benefit but ignore possible damage from potential unintentional species introductions. Several researchers and governments have recognized the importance of accounting for economic activity's environmental impact, called green national accounting (Fenichel & Abbott, 2014; Kubiszewski *et al.,* 2013). Progress in green national accounting has been seen in the Genuine Progress Indicator (Kubiszewski *et al.,* 2013), the Index of Sustainable Economic Welfare (Beça & Santos, 2010; Stockhammer *et al.,* 1997) and the Gross National Happiness measure (Ura *et al.,* 2012). Nonetheless, most of these green national accounting measures (aside from Beça & Santos, 2010) continue to ignore invasive alien species, which is a significant oversight. According to a recent report from the CBD on the resources needed to implement the Kunming-Montreal Global Biodiversity Framework, the cost of the continuous management of alien invasive species is estimated at $36 billion to $84 billion per year, depending on the assumptions used in the calculations (CBD, 2021b). To halt and reverse the trends of biodiversity loss and impacts on good quality of life, it is urgent to make the case for the importance of invasive alien species in the larger context of global biodiversity change (Mooney & Hobbs, 2000); cross-sector policy, coordination and collaboration have been identified as essential to invasive alien species prevention and control (**sections 6.2** and **6.3**).

Identifying financial and economic mechanisms to address invasive alien species is challenging for three principal reasons:

1. they affect public goods, which complicates the important task of defining responsibilities (Perrings *et al.,* 2002; **section 6.3.1.1**);

2. many of the costs and benefits of investment in invasive alien species management are non-market values (Perrings *et al.,* 2010), in some cases affecting values that cannot be monetized thereby limiting their consideration in economic flows and return on investment analyses (Auerbach *et al.,* 2014), and their use as arguments for generating resources and investment;

3. the ambiguous property rights of some goods and services that are affected by invasive alien species make it extremely difficult to implement public policy, legislation and regulatory mechanisms to protect these goods and services (Reichard *et al.,* 2005).

These three characteristics, compounded by the probabilistic nature of a successful invasion event (Fournier *et al.,* 2019) and the lag time that often separates an introduction from a successful invasion (Essl *et al.,* 2011), make it difficult to internalize the effects of invasive alien species, and therefore argue successfully for investing the resources necessary to adequately confront this global issue in a given region.

This IPBES assessment of invasive alien species cannot offer a comprehensive global review of existing financial and economic mechanisms, nor can it define a road map for success in the management of biological invasions (**section 6.2**). Rather, it presents some of the economic instruments available to finance different aspects of the invasion process, including prevention, eradication, containment, mitigation and restoration. The IPBES assessment of invasive alien species also examines some of the challenges, benefits, appropriateness and implications of adopting these instruments in different contexts and at various scales. It provides some insights for generating the economic incentives and deterrents that support a sustainable global effort to address the problem of invasive alien species in a more coordinated and better-financed manner. Some of the options presented are likely to resonate better in some regions than in others, and work best at some scales rather than others; the instruments outlined here are scale and context dependent. This simply reflects the diversity of the planet, its human societies and political systems, as well as the tremendous complexity of biological invasion process and its relationship with global biodiversity change.

Finally, there is no doubt that the power to advance the global invasive alien species agenda lies primarily in the hands of governments that lead legal and regulatory initiatives, supported by economic command-and-control instruments, tariffs and penalty systems. Government agencies are, and will likely continue to be, the organizations with the greatest capacity to respond to invasive alien species (Leadley *et al.,* 2014). However, in many cases, especially in countries with developing economies, multilateral and bilateral development aid will play a significant role. The magnitude and intrinsic characteristics of the problem call for an urgent and coordinated diversification of financing options and mechanisms.

6.5.1 Government financing

Government financing continues to play a leading role in invasive management efforts. However, this varies greatly between regions and countries (**Figures 6.4** and **6.5**), partly because of specific national fiscal and regulatory policies and public sector development strategies. In many countries there is considerable government investment in management of biological invasions through sub-national and national support programmes. However, these rarely translate to a multilateral coordinated effort required to adequately address the problem (Tollington *et al.,* 2017). Surveillance and monitoring are also aspects that receive government funding through the efforts of different agencies, but most of these activities have short time horizons, are limited to a few species and lack coordination, which all work to diminish their impact over time (Liebhold *et al.,* 2021).

To achieve adequate management of public funds to respond to invasive species, it is useful to coordinate robust policies at all levels of government in which diverse areas of administration are involved, including but not limited to financial, economic and environmental regulatory bodies, as well as those in charge of international trade and commerce and foreign policy (Tollington *et al.,* 2017, **sections 6.2** and **6.3**). Likewise, it is important to finance educational and outreach strategies to gain public support for the investment of resources into prevention, control and eradication projects (Bertolino & Genovesi, 2003).

Dividing the main sources of tax revenue into four broad categories allows us to understand how each could provide opportunities to respond to invasive alien species at different stages of the biological invasion process:

Direct taxes

The first group is direct taxes paid by households and businesses, which include income taxes, payroll taxes and corporate income taxes, among other taxes (i.e., capital gains and other investment incomes). A portion of direct taxes, which in theory are a reliable source of tax revenue, could be redirected to invasive alien species projects. Specifically, direct taxes are best used to address pre-invasion stages. In pre-invasion stages, biological invasions can be prevented with constant vigilance, including surveillance and monitoring, and therefore sustained funding. Also, investments made in preventing invasions pay dividends, as they eliminate invasions and all the associated costs.

Indirect taxes

The second category comprises indirect taxes, or taxes paid to the government or other public body through a third party, such as a retailer or suppliers. This category includes value added taxes, sales taxes, special taxes on products such as alcohol or tobacco and import duties. Indirect taxes are slightly less dependable than direct taxes because they vary with both household and commercial consumption patterns, as well as the specific tax policies of a given jurisdiction within a nation. Additionally, some subcategories of indirect taxes such as excise taxes can place an unequal burden on taxpayers at different income levels, as the tax per unit of a given good or service will constitute a higher portion of a lower-income taxpayer's income. However, the mandatory nature of this category of tax revenue means that they still are by-and-large a dependable source of tax revenue. Thus, they are appropriate sources of revenue for regular post-invasion control, mitigation and management programmes.

Non-tax revenues

In the third category are non-tax revenues from state-owned enterprises, including revenues from natural resources such as oil and gas. This category could help in research stages, as revenues from such sources already subsidize research in public universities and other institutions in some regions of the globe. Funding for research could be directed towards creating partnerships between public research institutions and natural resource management programmes to increase communication between academia and those responsible for implementing management programmes for biological invasions. This idea is particularly attractive as surveys of invasive alien species programme managers commonly cite a lack of communication with invasive alien species experts as a major barrier to implementing holistic invasive alien species management programmes (Beaury *et al.,* 2020).

External sources

Finally, there is funding from external sources, such as from bilateral or multilateral funding agencies (i.e., World Bank Group: International Bank for Reconstruction and Development (IBRD), International Development Association

(IDA), International Finance Corporation (IFC), Multilateral Investment Guarantee Agency (MIGA), International Centre for Settlement of Investment Disputes (ICSID), Asian Development Bank (ADB), International Monetary Fund (IMF) and International Fund for Agricultural Development (IFAD)), that are also considered public funding when the funds are disseminated through the recipient governments. Governments that depend heavily on funding from these sources are likely to not have other robust sources of tax revenue meant for controlling invasive alien species, and such funds therefore could be used to address the most pertinent areas of management of biological invasions in that region. In regions where there is significant government capacity for managing biological invasions, these funds could be directed specifically towards increasing international coordination of management efforts such as prevention and monitoring. External funding sources could support any action considered as a priority by the government, that requires special support and/or is very costly, such as eradication, or action that depend on international coordination, such as prevention and monitoring.

In summary, the coordinating role and much of the economic drive to address and reduce the risk of invasive alien species currently involves state actors. However, biological invasions are a complex problem with many facets and actors involved (**section 6.2**), so it cannot be thought of as the sole responsibility of governments. While fiscal policies and regulations at the national level have the potential to establish central guidance and coordination mechanisms, it is also important to manage multilateral and bilateral mechanisms, philanthropic support and, above all, to involve the private sector to reinforce government initiatives and address neglected aspects of the problem (Epanchin-Niell, 2017).

6.5.2 Laws, regulations and incentives for the private sector

Three examples of tools available to governments that incentivize the private sector to address invasive alien species prevention and control efforts include: ambient taxes, Pigouvian taxes and compensation, subsidies and fiscal incentives, and promoting the private sector to engage with prevention and control of invasive alien species.

6.5.2.1 Ambient taxes and subsidies

Beyond the more standard sources of tax revenue mentioned above are ambient taxes. Ambient taxes' purpose is to levy taxes on industries responsible for generating non-point sources of pollution, such as carbon emissions or invasive alien species. This type of taxation in the context of invasions, as introduced by Segerson (1988), would incentivize risk reduction by encouraging a shift towards more eco-friendly choices and ensure socially optimal behaviour in both the short and the long run (K. R. Jones & Corona, 2008). Furthermore, these taxes could serve as a cost recuperation strategy (to help pay for invasion impacts) and provide both the financial resources for prevention (including subsidies, surveillance and monitoring) and control strategies. Research has suggested that ambient taxes can be applied to users of ports to incentivize vessels to use proper, resource-appropriate, biosecurity measures (K. R. Jones & Corona, 2008). Effective ambient taxes are tailored to the nature and impacts of invasive alien species from specific ports of entry, which is achieved through greater levels of communication between regulators, researchers and industry stakeholders to reach appropriate tax rates. Furthermore, more research is needed to ensure that these taxes are levied without placing an undue and unequal burden on actors in international trade and into how to properly value the impacts of invasive alien species (Epanchin-Niell, 2017; K. R. Jones & Corona, 2008).

6.5.2.2 Pigouvian taxes

Pigouvian taxes, also known as an "introducers pay" tax, are another market-based approach to addressing invasive alien species control and are a particularly important policy tool when the private control of invasive alien species introductions is insufficient because of negative externalities or impacts from introductions that extend outside the market. Pigouvian taxes aim to tax individuals or companies to interiorize external costs not included in the market price (Sandmo, 2008). The overall aim is to incentive invasion prevention by inducing a cost to the expected damages from an invasive alien species to a level that equals the marginal cost to producers of reducing the risk of invasions. By doing this, producers would interiorize the societal costs not usually included in the market price (i.e., externalities). These taxes are usually perceived as a socially efficient strategy for reducing invasion risk (Epanchin-Niell, 2017), as these taxes force producers to account for the costs accrued to all of society from the risks of possible invasions. However, it is essential to notice that for Pigouvian taxes to yield qualitatively socially desirable behaviours, revenues from these taxes need to be used to offset the impacts caused by the introduced species (Fenichel *et al.,* 2014; Sandmo, 1975). Pigouvian taxes could be considered for imported goods or for dealing with neighbouring or spatial spillover effects (McDermott *et al.,* 2013). In some instances, Pigouvian taxes may even be more efficient than other market-based approaches like ad valorem taxes or tradable permits (McDermott, 2015; Richards *et al.,* 2010). However, like all biological invasions policies, a one-size fits all approach is not recommended and would require significant evaluation and consideration before implementation (Fenichel *et al.,* 2014; Knowler & Barbier, 2005; McAusland & Costello, 2004).

6.5.2.3 Leveraging compensation, subsidies and fiscal incentives and mechanisms

The widely used mitigation hierarchy framework (BBOP, 2012; IFC, 2012) establishes compensation mechanisms in cases of unavoidable and irreparable damage to biodiversity (Arlidge *et al.,* 2018). In instances involving damage by invasive alien species, these measures could be associated with robust and transparent mechanisms for monitoring, regulation and planning, as well as the creation of legislation that provides information on processes and responsibilities. In the case of subsidies, especially in the agricultural sector, these measures could likewise be well legislated and regulated as they can push producers to focus on improving production through reducing the use of unsustainable practices such as monocultures and excessive use of pesticides that can reduce the resilience of ecosystems to possible invasions (OECD, 2017; Robin *et al.,* 2003). It is therefore important to promote interdisciplinary research to develop evaluation mechanisms and indicators that help to anticipate the unexpected effects that these mechanisms may have at different scales and in different agricultural production modalities. In relation to fiscal and economic incentives (Fernandez, 2011), these could not only be oriented to reduce risk, but also to increase the resilience of ecosystems and social groups at high risk of invasions, i.e., the creation of fiscal and economic incentives that promote activities that help prevent, control, manage and eradicate invasions; but also, fiscal and economic incentives that discourage activities that promote the transport, introduction and establishment of invasions; but also, fiscal and economic incentives that discourage activities that promote the transport, introduction and establishment of invasions such as exotic gardens (Dutta *et al.,* 2021) and exotic pet trade (Gippet & Bertelsmeier, 2021).

6.5.3 Multilateral and bilateral financing organizations

Multilateral and bilateral funding organizations already support development and infrastructure programmes around the globe, but their resources and capacity to foster long-term change vary widely with both their organizational priorities and the regions in which they operate (Ray, 2021; **section 6.2**). However, there are opportunities for these organizations to partially redirect their efforts to support the invasive alien species problem without significantly altering their organizational priorities. One potential mechanism would be to update aspects of environmental impact assessments in development and infrastructure projects to place greater emphasis on invasive alien species. Environmental Impact Assessments already gather information on biodiversity (GBIF Secretariat & IAIA, 2020) that, with some effort and coordination, could be oriented to become mandatory, funded mechanisms that provide data for invasive alien species monitoring and prevention systems. Furthermore, environmental auditing has becoming an integral part of infrastructure and other developmental projects (W. Cook *et al.,* 2016). Inclusion of invasive alien species as an indicator in the environmental auditing of such developmental projects may indirectly fund the invasive alien species prevention and control activities. These same mechanisms could also assist in the selection of native species for restoration programmes.

While technical mechanisms are used by multilateral agency groups to incorporate environmental considerations into their investment portfolios, such as Performance Standard 6 (IFC, 2012), these standards do not directly address biological invasions. Though organizations such as the Equator Principles already work with development agencies to explicitly address biological invasions in their environmental considerations, it is also urgent to work with new multilateral and bilateral funding agencies that include large emerging economies countries such as Russia, China, India, Brazil and South Africa. Furthermore, while national governments could require multilateral funding organizations to carry out long-term monitoring of the impacts of development projects, this is often difficult in practice due to the limited capacity of some governments. Therefore, it is important that multilateral funding organizations include in their priorities and budgets adequate resources for sufficient invasive alien species monitoring and evaluation processes.

In order to help to ensure the necessary precautions are taken in the investment portfolios of multilateral organizations, insurance companies and financial institutions could be required to invest in modelling and managing the risks associated with invasive alien species within their various investment activities. Due to the sheer scale of global capital invested in transportation, infrastructure, energy, extractives and other development activities, modelling and management of invasion risk presents an opportunity to prevent negative impacts before they occur. However, for this to be effective it is necessary to develop instruments that interiorize the externalities of societal and environmental impacts of invasions (**section 6.5.2.2**). It is important that these agencies also create funding mechanisms to support research, monitoring and the creation of indicators by sectors such as non-governmental organizations and academia.

Finally, it is important to mention a recent movement to reduce funding for programmes that focus solely on increasing agricultural production and redirect that funding to develop the circular economy (**Glossary**) and incentivize bio-economic considerations and regenerative agriculture activities (Geng *et al.,* 2019). This could become an opportunity to broaden the investment portfolio of multilateral organizations to support countries in their efforts to establish taxes and fees that benefit invasion prevention and mitigation activities indirectly through strengthening ecosystem resilience to invasive alien species.

6.5.4 Private sector

Investment risk and firm reputation are two important factors driving the private sector (Kocovsky *et al.,* 2018). The private sector could increase its capacity to assess how investment decisions that maximize short-term economic returns also have the potential to trigger biological invasions that can have a devastating effect on its own finances in the medium and long run. As with multi- and bilateral funding organizations, mechanisms could be promoted to help the private sector include the invasion risk component in its economic analysis of different investment options.

To this end, it is important that governments develop and implement policies and legislation at the national and regional level that encourage private firms to include and disclose invasion risk in their reporting frameworks. These analyses can be supported by scientists and contribute to wider research on the development of analytical frameworks and risk indicators. Furthermore, large private sector companies with significant influence and investments in supply chains can help their respective industries to take the necessary considerations to reduce risk through introducing mechanisms that certify products (i.e., green labels), track origin, and generate freely available information to assess, anticipate and monitor the risk of invasion (Kotchen, 2013; Padilla & Williams, 2004). This latter approach could have the extra benefit of raising public awareness of the private sector's role in purposefully or inadvertently creating invasions, thereby making it more attractive for private firms and governments alike to invest in invasive alien species management to protect their public image (Hanley & Roberts, 2019).

Voluntary and self-regulating models, such as corporate social responsibility (CSR) strategies, can also be valuable tools for preventing biological invasions. These strategies imply companies are conscious of the realized and potential impact their activities have on all aspects of society, including economic, social and environmental (Lindgreen & Swaen, 2010). CSR strategies can take multiple forms. These can be voluntary programmes and partnerships to mitigate the environmental impact of industrial plants and production methods (Lindgreen & Swaen, 2010; Rondinelli & Berry, 2000). Alternatively, strategies can include the development of sourcing and marketing initiatives that protect social welfare and commit to environmental benefits (Lindgreen & Swaen, 2010; Roberts, 2003). Wildlife trafficking can be used as an example as to how the private

Box 6 17 Synergies with control mechanisms for illicit wildlife trafficking.

Illegal trafficking of biodiversity has been shown to be one of the main sources of invasive alien species in regions receiving illegally trafficked animals and plants (García-Díaz *et al.,* 2017). Efforts are underway to create new funding mechanisms and strengthen existing ones to combat illegal wildlife trafficking (Wright *et al.,* 2016). One element that can help to deter this illegal activity is the speed of response in relation to species identification (e.g., Kretser *et al.,* 2015). This can be achieved by supporting integration mechanisms of control systems at regional and global levels and increasing the response capabilities of regulatory institutions such as customs and migration agencies at ports of entry and exit (Fajardo del Castillo, 2016).

Previous studies of government response to invasive alien species have identified increased collaboration amongst countries as essential to any future management efforts for biological invasions (Hardisty *et al.,* 2019; Perrings *et al.,* 2010). One option is to develop software intended to foster communication networks (Wallace & Bargeron, 2014; Wise, 2019) and disseminate technical training between and amongst regulatory organizations at the international level (**sections 6.3.1** and **6.6**). However, the process of technification and delivery of capacity-building to regulatory entities and personnel is costly (e.g., Juffe-Bignoli *et al.,* 2016). One way to avoid placing the burden solely on state organizations would be to increase both the criminal and civil liabilities of international freight companies to incentivize those organizations to take the proper precautions to avoid those penalties. This would also help the economic sustainability of expert regulatory entities to maintain the employment of trained technical staff.

Although the private sector has great potential to leverage mechanisms and business practices to help with the issue of invasive alien species, this will not happen without the support of consumers willing to pay the premium for safe products (Akerlof, 1970; Cason & Gangadharan, 2002). Governments could also create mechanisms and conditions for the private sector to feel that it is profitable to invest in invasive alien species prevention, monitoring and the certification of processes and products that directly address the issue of invasive alien species. One way to do this is the promotion of codes of practice for the translocation and exploitation of invasive alien species (**section 6.3.1.3(4)**) and green labelling (**section 6.5.5**). Empowering consumers to exert pressure on large, multilateral corporations to make decisions that help with this problem such as marking products with certifications that consider biological invasion processes can be an acceptable mechanism for the private sector (Kotchen, 2013) that in turn will provide companies with the favourable standing needed to succeed in a competitive global market.

sector is stepping up to help end the illegal commerce of species (e.g., the United States Wildlife Trafficking Alliance) and the tools that can be used to prevent biological invasions **(Box 6.17)**. In this context, codes of practice (**section 6.3.1.3 (4)**) are a viable way by which the pledges made in corporate social responsibility (CSR) strategies can be translated into resources and actions to address the problem of biological invasions.

6.5.5 Role of global supply chains

The impact of global supply chains on the transport and introduction of invasive alien species is undeniable (Hulme *et al.,* 2018; Seebens *et al.,* 2017). The introduction and establishment of invasive alien species are closely related to international trade flows and global trade routes, with international shipping being the main vector for the introduction of invasive alien species (Seebens *et al.,* 2015; Westphal *et al.,* 2008). If the trend of global trade growth continues, it is estimated that the direct annual cost of management of biological invasions in 2050 could reach $36 to $84 billion per year (Deutz *et al.,* 2020).

Incentivizing changes in supply chain management practices offers the opportunity to strengthen the prevention of alien species introductions and therefore decrease the costs associated with controlling and eradicating invasive alien species. One of the key components in driving change in supply chain management practices is elevating the importance of invasive alien species in the minds of end consumers (Hanley & Roberts, 2019). The changes made would encompass corporate commitments to assessing and improving corporate policies, internal standards and funding mechanisms to ensure that supply chains take appropriate precautionary measures, especially in producer countries.

Investments can be made to both improve current practices and elevate the importance of invasive alien species – safe practices in the minds of those responsible for setting corporate strategy (Kocovsky *et al.,* 2018). Importing countries could collaborate with exporting countries to improve sustainable practices that reduce the probability of invasions through their integration into regulations and international trade agreements (**sections 6.3.1.3** and **6.3.2.2**). In this sense, the integration of the component of biological invasions in green labels and certification systems (e.g., Blackman & Rivera, 2010) is especially important because these systems have been shown to be effective in raising public awareness of issues such as deforestation, though labelling by itself does not directly decrease deforestation (van der Ven *et al.,* 2018). Increased public awareness of invasive alien species is a fundamental component in garnering support for new funding mechanisms and policies that have the potential to address invasive alien species more directly. These methods also transfer a large part of the decision to consumers, thereby increasing awareness of the invasive alien species among the public.

6.5.6 Role of philanthropy, non-governmental organizations and academia

While philanthropy represents a significant source of funding for environmental issues such as invasive alien species in some regions, it is almost non-existent in others due to prevailing economic and social systems at the national and subnational level. In some cases, foundations and their philanthropy programmes may be limited by their internally defined priorities, which in many cases are aligned with topics of more widespread public concern (E. R. Larson *et al.,* 2016; Macdonald *et al.,* 2017). This dynamic makes it difficult to develop far-reaching programmes in less visible, but nonetheless important aspects of the invasion process. On the other hand, philanthropic organizations also offer funds to explore innovative invasive alien species programmes (E. R. Larson *et al.,* 2016), but these are quite limited in scope and tend to be used to support specific efforts that align with larger strategic goals of the organizations that receive them. Philanthropic funds are perhaps best used to finance the development of tools and pilot projects that can act as proofs of concept for later implementation by larger, better-funded entities such as governments or financial organizations such as the Global Environment Facility (GEF). In the case of funds that come from corporate social responsibility programmes and multilateral corporations, one option would be to develop metrics and methodological frameworks that the private sector can integrate into their business models to help them report on the impact and investment risk of invasive alien species.

Non-governmental organizations benefit, in large part, from funding sources that have their origins in philanthropy. Although philanthropic organizations have internal mechanisms to define priorities, non-governmental organizations are more transparent in this sense and can channel different philanthropic funds and articulate them to coordinate with programmes pursuing the same objective. Non-governmental organizations, in their constant search for funding to sustain themselves, have the flexibility to change their strategic goals swiftly. This apparent flexibility of the non-governmental organizations can be seen as an asset, as it allows the adaptability of their programmes to be maintained over time; but it also has the potential to drive significant changes in their programmes, and even terminate them altogether.

Finally, there is academia, which also moves with funding from philanthropy, but also receives significant government funding in many parts of the world. The way in which lines of research are often established early in a researcher's career presents the opportunity to begin cultivating a new generation of invasive alien species specialists in diverse fields. Addressing biological invasions can be achieved through greater coordination between academia and those responsible for implementing invasive alien species best practices; therefore, investments in invasive alien species research can essentially be seen as investments in invasive alien species prevention. One example of this is the Global Register of Introduced and Invasive Species (GRIIS) – a collaborative output demonstrating best practice use of biodiversity informatics to make invasive alien species checklists open (Pagad *et al.,* 2018, 2022). While it is important that foundations provide the funding that supports non-governmental organizations and academia to generate the early ideas that catalyse larger efforts, these efforts could be connected to the private sector, multilateral banks, and the governments to move ideas from pilot projects and proofs of concept to established, long-term programmes. National strategies for invasive alien species are a central mechanism by which this connection can be enabled.

6.5.7 International funding

The mechanisms described here are not the complete solution to financing the global invasive species problem. However, these mechanisms can drive significant change if they are supported, enacted and implemented by governments, multilateral and bilateral organizations, multilateral development banks, philanthropic foundations, non-governmental organizations, academia and the private sector, in a coordinated manner with strong support from informed citizens.

All the options reviewed in the IPBES invasive alien species assessment could benefit from considering different socioeconomic and cultural realities, and presenting common and coordinated strategies that take into account the communities most affected by biological invasions. There are large differences between countries in their capacity to tackle the problem of invasive alien species (Early *et al.,* 2016) and a significant geographic bias in data availability regarding the invasive alien species (**Chapter 2**, **section 2.1.4**; **section 6.6.1(3)**). These limitations have hampered global efforts to reduce the introduction of alien species and prevent their impacts. Flow of financial and other resources from developed countries to developing countries, particularly in Asia and Africa, can improve the understanding of the complex phenomena associated with biological invasions and help developing countries in their efforts to prevention and control of invasive alien species.

Multilateral development banks are in a great position to lead change towards suitable development (Handl, 1998; Trillo, 2021) and achieving the targets set by the Kunming-Montreal Global Biodiversity Framework. At UNFCCC COP 26 in Glasgow, United Kingdom, ten multilateral development banks signed a joint statement on Nature, People and Planet (Messetchkova, 2021), which recognizes that "tackling global poverty, climate change, and the drivers of nature and biodiversity loss are inextricably linked and affirms their commitment to further mainstream nature into their policies, analyses, investments, and operations." under this banner, projects sponsored by these institutions could consider projects aimed at reversing the nature loss caused by invasive alien species.

While it is true that the CBD or the Global Environment Facility could serve to mobilize financial resources, the burden of financing a global strategy need not fall solely on governments and their fiscal policies. It is beneficial to involve all sectors and actors in order to expand the financial resources available. This could also increase the scope of public policies and private sector practices towards sustainability. The need to increase the level of financial resources from all sources and increase the availability of these resources for developing countries is embedded in Target 19 of the Kunming-Montreal Global Biodiversity Framework. Key to this effort will be framing these efforts as medium- and long-term investment opportunities, rather than as necessary sacrifices.

The report on the global biodiversity financing gap estimates that between 722 and 967 billion dollars would be needed to sufficiently confront the crisis, with invasive alien species alone representing between 36 and 84 billion dollars (Deutz *et al.,* 2020). However, these estimates have wide ranges of error due to the limited availability of global biodiversity indicators (Mcowen *et al.,* 2016), as well as many of the data gaps described elsewhere in this assessment. Many uncertainties are related to future investments and the different funding mechanisms that could directly or indirectly support efforts against invasive alien species, but two things are clear: it is possible to reduce the need to invest in the control of and increase investment in the prevention of invasive alien species.

In the era of climate change wherein there is a growing understanding of the interconnectedness of all human activities, both sustainable and unsustainable, it is also paramount not to ignore the ways in which invasive alien species and efforts to combat them might influence and be influenced by other conservation efforts. For example, an important source of funding for biodiversity conservation in general is the carbon credit market, wherein governments voluntarily create offset mechanisms for sustainable forestry practices. Although this funding does not explicitly target biological invasions, establishing transnational safeguards in relation to reforestation and other restoration practices will help to quantify the contribution of carbon credit markets financial mechanisms to preventing invasions.

6.6 INFORMATION OPTIONS

Knowledge of invasive alien species is deeply embedded in the knowledge of the natural world, such as how organisms live, reproduce, disperse and interact. This knowledge is in turn disseminated as information, in different languages, cultures, media and disciplines. Much of it is not permanently preserved, either because it is experience in the minds of practitioners or because it is documented on temporary media. Some information, particularly from scientific publications, is available only from specialized libraries and databases or only at great expense or in a limited number of languages (Nuñez & Amano, 2021). Other knowledge, for example, of pastoralists, is passed down orally between generations and is not necessarily documented. Some knowledge has been rigorously tested using the scientific method, whereas other knowledge is based on observations or on a belief system (Shackleton, Richardson, *et al.,* 2019).

Even if invasive alien species knowledge were all documented and adequately archived there are problems associated with delivering this knowledge to the people who need it. For example, alien species are, by definition, remote from their origins. In the initial stages of an invasion, knowledge of the invader is likely to be better in its native range, or in previously invaded areas, than in the newly invaded range. This disparity includes both access to written knowledge and communication with practitioners who have experience of the invader. Thus, much knowledge that exists on invasive alien species is not adequately findable.

Invasive alien species span the full taxonomic range of species, from large mammals and trees to protozoa and algae (**Chapter 1**, **section 1.3.1**). It is therefore hard to generalize about the information required to support policy on invasive alien species. Knowledge is required both in depth and breadth. That is to say, detailed information on some invasive alien species can provide strong evidence for policy decisions. Yet a broad overview of all alien species would be needed to foresee future threats and to understand the impact of invasive alien species on other species and on people. This makes prioritization of knowledge acquisition difficult, particularly in view of the level of uncertainty in the threats.

Much of the information on invasive alien species is provided by general sources of biodiversity knowledge (Ramírez-Albores *et al.,* 2019). Knowledge sources specific to invasive alien species are also available (Ricciardi *et al.,* 2000), but are restricted to those species known to be alien. In both cases, these sources are often nationally or regionally circumscribed and created for local readers in their own language.

This section first broadly summarizes the knowledge needs identified by previous chapters of this assessment (summarized in **Table 6.10**; **Supplementary material 6.2**), and then discusses key options (**section 6.6.2**) for strengthening the generation and flow of policy and management-relevant invasive alien species information. It also introduces the particular problems faced by Indigenous Peoples and local communities, or isolated communities.

Table 6 10 **Cross-chapter synthesis of gaps in data, information, knowledge and understanding.**

Category	Gap
Gaps on biomes, units of analysis and species groups (section 6.6.1.1)	Incomplete or lack of inventories of invasive alien species in marine, tropical and Arctic ecosystems (**Chapter 2**, **sections 2.5.2.1**, **2.5.2.4**, **2.5.2.5**, **2.5.4**)
	Incomplete or lack of inventories of invasive alien microorganisms and invertebrates (**Chapter 2**, **sections 2.3.1.11**, **2.3.3.3**)
	Lack of understanding of the drivers facilitating biological invasion for some animal groups (notably invertebrates), fungi and microbes (**Chapter 3**, **section 3.6.1**)
	Lack of understanding and synthesis of the impact of invasive alien microbes (**Chapter 4**, **section 4.7.2**)
	Poor understanding of drivers facilitating biological invasions in aquatic and marine systems (**Chapter 3**, **section 3.6.1**)
Regional gaps in data and knowledge (section 6.6.1.1)	Comparatively incomplete inventories of invasive alien species in Africa and Central Asia (**Chapter 2**, **sections 2.4.2.5**, **2.4.5.5**)
	Comparative lack of understanding of the drivers facilitating biological invasions in developing economies (**Chapter 3**, **Box 3.12**)
	Lack of data and knowledge of the drivers facilitating biological invasions in sub-Saharan Africa, tropical Asia and South America (**Chapter 3**, **section 3.6.3**)
	Incomplete data on the impact of invasive alien species across Africa and Central Asia (**Chapter 4**, **section 4.7.2**)

Table 6 10

Category	Gap
Interoperable data for monitoring and research on invasive alien species and on the effects of drivers of biodiversity change **(section 6.6.2)**	Lack of standardization of terminology for invasive alien species monitoring (**Chapter 2**, **section 2.4.4.5**; **Chapter 6**, **sections 6.6.2.3**, **6.6.2.7**)
	The drivers facilitating biological invasions for some animal groups (notably invertebrates) and in fungi and microbes are poorly understood (**Chapter 3**, **section 3.6.1**)
	Lack of information on the role of indirect drivers, especially governance and sociocultural drivers, in affecting biological invasions (**Chapter 3**, **section 3.6.1**, **Box 3.12**)
	Lack of understanding of the net effects of multiple interacting drivers in shaping and promoting biological invasions (**Chapter 3**, **section 3.5**, **Box 3.10**, **section 3.6.1**, **Box 3.13**)
	Lack of knowledge on interactions and feedbacks across drivers in promoting invasions (**Chapter 3**, **section 3.6.3**)
	Lack of integration of data and knowledge sources on impacts across languages (**Chapter 4**, **section 4.7.2**)
	Incomplete data to undertake risk management, cost-effective species-led surveillance and detection of fungi, microbes and marine pests (**Chapter 5**, **Table 5.11**)
	Incomplete data to prioritize biological invasion management under climate, sea- and land-use change (**Chapter 5**, **section 5.6.1.3**)
	Lack of inventories at fine scales and for specific taxon and biome contexts to support decision makers in determining when to implement species-led and site-based management (or both) (**Glossary**; **Chapter 5**, **sections 5.6.2.1**, **5.7**)
	Incomplete data to develop pathway risk assessments and management for different taxonomic groups and biomes (**Chapter 5**, **Table 5.11**, **section 5.6.2.5**)
	Incomplete data and understanding of site-based and ecosystem-based management concepts (**Chapter 5**, **section 5.6.2.1**)
	Incomplete data and understanding of the conditions that facilitate successful integration of policy developments into management plans (**section 6.6.1.4**)
	Lack of indicators[14] of the various dimensions of biological invasion that are policy-relevant, sensitive, reliable, relevant at national and global scales, sustained for medium-to-long-term tracking of progress and part of a responsive policy environment (**section 6.6.3**)
Gaps on how invasive alien species affect Nature's contributions to people **(section 6.6.1.4)**	Incomplete data on impact on nature's contributions to people and good quality of life (**Chapter 4**, **section 4.7.2**)
Management and policy approaches **(section 6.6.1.2)**	Lack of control options for marine invasive alien species and invasive microbial fungal pathogens of plants and animals (**Chapter 5**, **section 5.6.1.1**)
	Lack of agreed-upon methods of supporting management decision-making for invasive alien species with both positive and negative impacts (**Chapter 5**, **section 5.6.1.2**)
	Lack of methods of managing pathways for invasive alien species arriving as contaminating invasive alien species, or through shipping containers, e-commerce (legal/illegal), biofouling or ports, and across land borders and along trade supply chains (**Chapter 5**, **Table 5.11**, **section 5.6.2.4**)
	Lack of methods for adaptive management of invasive alien invertebrates and plants using alternative approaches given the declining number of chemical control options (**Chapter 5**, **section 5.6.2.5**)
	Lack of eradication guidelines and strategies for generalist invasive alien invertebrates, diseases and hard-to-detect freshwater and marine invasive alien species (**Chapter 5**, **section 5.6.2.1**, **Table 5.11**)
	Missing information on the implementation of adaptive-collaborative governance for biological invasions and factors important for the success of this governance strategy (**section 6.4.4.4**)
	Incomplete data on the effectiveness of policies, management strategies and actions related to biological invasions (**section 6.6.3**)
	Lack of scenarios and models of invasive alien species that consider interactions with other drivers of change in nature (**Chapter 2**, **section 2.6.5**; **Chapter 6**, **section 6.6.1.6**)
	Lack of biological invasion research that includes social dimensions to generate socially relevant additional data and knowledge, better inform management and policy and build trust between sectors of society (**sections 6.4**, **6.6.1.4**)

Table 6 10

Category	Gap
Management and policy approaches **(section 6.6.1.2)**	Lack of multidisciplinary to interdisciplinary research on policy regimes and governance for biological invasions (**sections 6.2.4**, **6.5.1**)
	Lack of tools and frameworks to predict biological invasions (**sections 6.2.1**, **6.6.1.6**, **6.7.2.7**)
Gaps to fill to support the implementation of policy and management **(sections 6.6.1.2, 6.6.1.3, 6.6.1.6)**	Lack of tools to reduce the barriers to information-sharing within and across countries (**section 6.6.2**)
	Lack of research and data on how best to implement context-specific integrated governance systems to manage biological invasions (**sections 6.6.1.3**, **6.6.1.4**, **6.6.2**)
	Lack of mechanisms that allow effective collaboration among different aspects of the socioecological systems (**Figure 6.7**, **section 6.7**)
	Policy for new and emerging technological innovations for invasive alien species management to support effective development and implementation and prevent or manage risks (**section 6.3.3**)
	Additional, particularly fine scale, data on how invasive alien species are introduced and spread to support prioritization of introduction pathways and pathway management (**Chapter 2**, **section 2.1.2**; **Chapter 5**, **section 5.6.2**; **Chapter 6**, **section 6.6.1.2**)
	Research and design of economic options, including the tailoring of ambient taxes and analyses and indicators to assist private companies (**section 6.5.1.1**)
Knowledge gaps on invasive alien species of particular relevance to Indigenous Peoples and local communities **(section 6.6.1.5)**	Lack of information on invasive alien species status and trends on land and water managed by Indigenous Peoples and local communities (**Chapter 2**, **Box 2.6**)
	Lack of clarity on how knowledge, resources and data on invasive alien species should be treated under the Nagoya Protocol (**section 6.6.1.5**)
	Mechanisms for sharing knowledge on invasive alien species with Indigenous Peoples and local communities (**section 6.6.1.5**)
	Understanding the on-the-ground experiences of stakeholders and Indigenous Peoples and local communities and their engagement in invasive species management and governance (**section 6.4.1**) and related network analysis (**section 6.4.4.4**)

14. A headline indicator has been proposed for planning and tracking of progress towards target 6 of the Kunming-Montreal Global Biodiversity Framework, with opportunities to build on existing indicators for biological invasions (**section 6.6.3**).

6.6.1 Invasive alien species information needs

Knowledge gaps result from extreme heterogeneity in the collection and distribution of information and data. Given limited resources so-called gaps could therefore be defined by the questions and the problems that need solutions.

There are many unknowns about the biology of invasive alien species. These are, in part, known limits to what one knows about these species **(Box 6.18)**. Such limits can be described in terms of expressions of uncertainty or as knowledge gaps. The most problematic cases are those species that are entirely unexpected when they start to invade (so-called "unknown-unknowns", or "ignorance" in **Figure 6.18**; Taleb, 2007). Nevertheless, such cases may be novel only to certain sectors and locations. Therefore, inter- and intra- sectoral communication is essential to ensure that the number of surprises (unexpected cases) are minimized. Without such communication it is unrealistic to expect actors in policy and management of biological invasions to be adequately prepared.

Table 6.10 presents a synthesis of knowledge gaps identified in the IPBES invasive alien species assessment. Some of the knowledge gaps are relevant globally, for example the need to increase understanding of the outcomes of multiple interacting indirect and direct drivers of change in nature. Others apply to specific nations or regions and highlight the potential to improve the information and data flow from some regions. There are also gaps in the understanding of the interplay between social, economic and environmental factors that link policy and governance structures. These gaps are best perceived as opportunities to embrace emerging tools and technologies to underpin decision-making and management of biological invasions; indicators and targets on invasive alien species will benefit from improved scenarios and models which are currently limited by the knowledge gaps outlined in this chapter.

Box 6 18 **A case that illustrates the problems of unknowns in knowledge dissemination is the spread of ash dieback disease in Europe.**

This fatal disease of *Fraxinus excelsior* (ash) was detected in Europe the mid-1990s. It was then described as a new species *Chalara fraxinea* (Kowalski, 2006). It subsequently spread across the whole of the European range of *Fraxinus excelsior* (ash). In 2009 ash dieback was identified as being the anamorph of *Hymenoscyphus albidus* that had been described from Europe in 1850 and was apparently native (Kowalski, 2006). However, it was subsequently realized that *Chalara fraxinea* and *Hymenoscyphus albidus* are two cryptic species largely indistinguishable morphologically (Queloz *et al.,* 2011). One causes the pathogenic disease of ash and the other is a harmless saprophyte. This determination led to the establishment of yet another name, *Hymenoscyphus pseudoalbidus*. However, it was later found that *Hymenoscyphus pseudoalbidus* was conspecific with Japanese specimens named *Lambertella albida* (Zhao *et al.,* 2013). Finally, due to the nomenclatural rules of priority and recent changes in the Code of Nomenclature for algae, fungi and plants, the name was changed to *Hymenoscyphus fraxineus* (ash dieback; Baral *et al.,* 2014).

It took twenty years since ash dieback was first discovered in Europe for a stable name for it to be arrived upon, making it possible to connect the species to what is thought to be the native range in Eastern Asia. It is difficult to know how much this confusion over the origin and name of this species contributed to a slow response to the spread of the disease and how much this has obstructed research. This example illustrates the different types of uncertainty associated with biological invasions, in this case both taxonomic uncertainty as well as the need for more research on the distribution and identity of this species group. Furthermore, in the case of ash dieback, once its origins were revealed it allowed information to be brought together from distant sources in time and space.

6.6.1.1 Biodiversity information needs

All seven of the general types of biodiversity knowledge shortfalls (Hortal *et al.,* 2015) are equally relevant to knowledge on invasive alien species and the information needed to resolve this problem, i.e., taxonomy, distribution, populations, evolution, traits and functions, tolerances and ecological interactions. The section below details these types of information needs for policy support on biological invasions and discusses current sources of information and how these are created and disseminated.

(1) Addressing taxonomic biases in research

Taxonomic bias is pervasive in knowledge of biodiversity (Haque *et al.,* 2020; Zamora-Gutierrez *et al.,* 2019), conservation sciences and practices (Creighton & Bennett, 2019) and ecological research (Rosenthal *et al.,* 2017), and such biases have not changed over time (Creighton & Bennett, 2019; Rosenthal *et al.,* 2017; Troudet *et al.,* 2017; **Chapter 2**, **section 2.3.1.11** for an example of information gap on animals). There are probably a number of causes for this. When compared, invasive alien species were more likely to be studied than non-invasive naturalized species (Pyšek *et al.,* 2008), although for many taxonomic groups invasive alien species are also poorly investigated. Other factors that drive taxonomic biases in research include societal preference, research funding, conservation policy (Jarić *et al.,* 2019; Troudet *et al.,* 2017) and probably also research tractability of the species.

Pauchard *et al.* (2011) found that the principal focus of invasive alien species publications in Latin American and Caribbean countries was introduced animals (65 per cent, 119 articles), and often the more tractable or emblematic species. The most studied aquatic alien taxa in South America were fish (26.8 per cent) and molluscs (25.2 per cent), followed by crustaceans, algae, cnidarians, polychaetes and ascidians (Schwindt & Bortolus, 2017).

Taxonomic biases limit the ability to understand the complex processes and interactions that underlie biological invasions. The information obtained from the study of single taxonomic groups is not necessarily transferable to others, for example, due to differences in impacts and dispersal pathways (Jeschke *et al.,* 2012). Therefore, studies targeting few species render generalizations either inaccurate or incomplete (Jeschke *et al.,* 2012). In invasion biology, few studies have examined failure of invasions (Pyšek *et al.,* 2008). However, studying both biological invasion success and failure is important to test invasion hypotheses and understand the overall process of biological invasions (Zenni & Nuñez, 2013; Diez *et al.,* 2009).

Given that taxonomic bias is recognized, any effort to minimize this shortfall will improve information of biological invasions and produce better informed management and policy decisions (Pyšek *et al.,* 2008). Strategies to reduce taxonomic bias include advertising poorly documented and under-studied species among professional and citizen scientists (Troudet *et al.,* 2017), and promoting cross-taxonomic studies that involve a set of invasive alien species that belong to different taxonomic groups (Jeschke *et al.,* 2012).

Tractability of collecting and processing species occurrence data is likely to be a component of these research biases.

Box 6.19 Genetic tools for detection, characterization and traceability of marine and aquatic invasive alien species.

The management of biological invasions can be improved through accurate identification of species to connect with information on their natural history and ecology. Traditionally, these species were identified using methods that require direct observation, or occasionally tracks and signs. In marine and aquatic ecosystems this is particularly problematic due the inaccessibility of working in much of the habitat. Genetic characterization provides an accurate molecular identification of the species and generates information to parameterize population models, genetic relationships, connectivity among populations and the effective population size (Díaz-Ferguson & Moyer, 2014; Díaz-Ferguson & Hunter, 2019; Estoup & Guillemaud, 2010). Molecular genetics can be used to detect founder effects, bottlenecks and hybridization processes that can occur during invasion (Roman & Darling, 2007; Frankham *et al.,* 2010). Genetic approaches can be used to answer questions such as: Are invasive alien species present in an area or region (application challenged in areas where information on native biota is incomplete, e.g., deep sea)? How many organisms are present in an area? Are these organisms able to reproduce? Where are the source populations of these organisms? (A. Barbour *et al.,* 2010; Estoup & Guillemaud, 2010).

For example, identification, genetic characterization and tracking of invasive alien species is only possible due to the development of genetic markers (Pochon *et al.,* 2013). A genetic marker is a deoxyribonucleic acid (DNA) target sequence used for molecular identification of a species or to determine its variability (Díaz-Ferguson, 2012). Since the advent of polymerase chain reaction (PCR; **Glossary**) and quantitative PCR methods several markers have been developed to identify, track and characterize the spatial variation of marine and aquatic populations including invasive alien species (Hulata, 2001; Féral, 2002). More recently the use of mini barcoding and quantitative PCR detection of environmental DNA allows scientists and managers to detect fragments of DNA left behind by species in non-living components of the environment (i.e., soil, sediments and water) without the need to observe or collect the focal species (Díaz-Ferguson & Moyer, 2014). Environmental DNA, although still developing as a technology to narrow uncertainty, has been demonstrated to be efficient at detecting species with a small population size such as invasive alien species in the course of establishment, or imperilled, threatened and endangered species (Jerde *et al.,* 2011). Marine ecosystems are just one of the areas where environmental DNA surveys are likely to radically change the detection and monitoring of invasive species (Chown *et al.,* 2015; Darling *et al.,* 2017; Holman *et al.,* 2019). These methods and others are covered in more depth in **Chapter 5**, **section 5.4.4.2**.

Next generation sequencing (environmental DNA; **Box 6.19**), machine observations (e.g., camera traps and space-based remote sensing) and machine learning will likely make new taxonomic research more feasible.

(2) Overcoming gaps in impact analysis

The number of alien plant species worldwide has been estimated to be in the thousands, but in 2013, robust impact studies were only available for fewer than 200 species (Pyšek *et al.,* 2013; **Chapter 4**). For example, information on the impacts of alien species on biodiversity and on Indigenous Peoples and local communities is a key gap, with particularly acute gaps on impacts across Africa and Central Asia and at the ecosystem level (**Chapter 4**, **sections 4.6.4** and **4.7.2**; **Table 6.1**). Even more substantial information gaps occur in the marine realm where only a small proportion of organisms have been evaluated for their impact in their non-native range, particularly in the deep sea and pelagic open ocean. In a meta-analysis of the impacts of invasive macroalgae data on only 12 species were found, of which only eight had experimental evidence of impact (Maggi *et al.,* 2015). Another review of marine aliens in Europe found only 13 per cent of the reported impacts were supported by experiments and most were only inferred from abundance of the alien and co-occurrence with potentially impacted native species (Katsanevakis *et al.,* 2014). See **Chapter 4**, **section 4.7.2**, for more details on the data and information needs to understand impacts and **Chapter 5**, **section 5.2**, for other use in decision-making.

(3) Reducing geographic bias in research

Invasive alien species are pervasive, but there is disparity of data availability and research efforts across geographic regions. All the studies that examined geographic patterns of data availability (i.e., species occurrence data) and research effort (i.e., publication efforts, ecological study sites) showed geographic biases with a general pattern of high data availability and research efforts in Europe, Australia and North America and low availability in Asia and Africa (Boakes *et al.,* 2010; Yesson *et al.,* 2007; **Chapter 2**, **Figure 2.6**). Such geographic bias is also prevalent among scenarios and modelling studies related to invasive alien species (**Chapter 1**, **section 1.6.7.3**). Geographic bias has already been highlighted in the other chapters (particularly **Chapter 2**, **Figure 2.6** and **section 2.4**; **Chapter 4**, **section 4.7.2**; **Box 6.18**). The recent publication of national checklists of invasive alien species for most of the world's countries helps to overcome this geographic bias, with the focus of these checklists being on invasive alien species with biodiversity impacts and not on all alien species (Pagad *et al.,* 2022).

A systematic review investigated invasive alien species in natural ecosystems (Lowry *et al.,* 2013) and found that such studies were mostly concentrated in North America, Western Europe, Eastern Australia, New Zealand and Hawaii, while there was a dearth of studies in countries located in the tropics, such as in Asia, Africa and Central and South America. This pattern is close to the geographic distribution of sites of overall ecological studies in terrestrial systems (L. J. Martin *et al.,* 2012). Nearly three-quarters of field studies have been done in terrestrial systems, with freshwater and marine ecosystems significantly underrepresented in studies of natural ecosystems (Lowry *et al.,* 2013; **Chapter 2**, **section 2.5.1**). Similarly, countries with a high percentage of IUCN Red Listed vertebrate species that are threatened by invasive alien species (e.g., Mexico, Colombia, Peru, Argentina, Madagascar, India, Indonesia) have a low percentage of publications on biological invasions (Bellard & Jeschke, 2016). In contrast, the countries with a low percentage of invasive alien species-threatened Red Listed species (e.g., Canada, United States, China) have high publication efforts on biological invasions. Alien birds, for example, have been reported in 247 regions across the world, but environmental impact data is available for only 60 regions (24 per cent; T. Evans & Blackburn, 2019).

Taking the example of Latin American and Caribbean countries, the number of articles on invasive alien species over the last 20 years was 344, with an increase after 2003 and a higher percentage between 2003 and 2008 (Pauchard *et al.,* 2011). The country with most articles on invasive alien species was Argentina (105), followed by Brazil (85), Chile (53) and Mexico (41). These four countries contributed 82.5 per cent of all the articles on invasive alien species from Latin American and Caribbean countries. Differences among countries reflect the asymmetry in invasive alien species research among the Latin American and Caribbean countries, but also the effect of country size. Most countries on the continent began publishing on invasive alien species only in the 1990s (Speziale *et al.,* 2012). However, the differences among countries in research effort on alien species does not seem to be just a matter of research budgets, nor differences between developed or developing countries, nor differences due to their higher biodiversity and the interest in protecting it. Although an explanation might be a lower number of invasive alien species in South America, scientific information to properly assess this remains lacking (Speziale *et al.,* 2012).

Generating timely and adequate information across geographic regions is an opportunity for implementing effective management strategies. This is particularly important for the aquatic realm, where eradication and control efforts are viable only at the very initial stages of the invasion process (Lehtiniemi *et al.,* 2012). Lack of information in regions highly vulnerable to invasive alien species may result in a delayed response to invasive alien species at an early stage (Bellard & Jeschke, 2016; **Chapter 5**, **section 5.6.2**). As a consequence of a lack of monitoring, occurrences of invasive alien species can remain unnoticed, thereby reducing the chances of early detection and eradication.

Low research investment and data availability in certain regions, such as parts of Asia and Africa (Bellard & Jeschke, 2016; T. Evans & Blackburn, 2020; Lowry *et al.,* 2013; Pyšek *et al.,* 2008), can mean that these regions are less understood and thus underrepresented when frameworks and theories are developed for biological invasions and their management. Therefore, collaborations between invasion scientists in developed and developing countries, and developing research capacity in less developed countries improves data availability for better understanding of the processes associated with biological invasions (Bellard & Jeschke, 2016). Since geographic biases are also apparent in authorship (corresponding author) of the research articles published in journals like *Biological Invasions*, with disproportionately high submission by authors from North America, Europe and Australasia, such biases can be minimized by encouraging manuscript submissions from countries of other regions (Nuñez *et al.,* 2021). Owing to a lack of study or expertise, discovery of invasive alien species in invaded areas can lag by decades or longer. The numbers of recorded marine invasive alien species are, for example, particularly likely to be underestimated. The size of this gap is difficult to assess, and it varies among different taxa, habitats and regions. Information is most accurate for large, conspicuous, multicellular organisms (Galil *et al.,* 2014; Ojaveer *et al.,* 2015).

(4) Invasive alien species – native and invaded ranges

There are significant data gaps on the spatial delimitation of the edges of species native and invaded ranges, particularly at scales fine enough to inform management decisions (Hardisty *et al.,* 2019; Latombe *et al.,* 2017). Species distribution, specifically native and invaded range, are derived from a wide variety of sources. Indeed, how the definition of what constitutes a native or alien species varies globally depending on the history of human migrations (**Chapter 1**, **Figure 1.1**, **sections 1.3.1** and **1.5.2**; Carthey & Banks, 2012). These differences in definition influence the scope of invasive alien species policy and although the differences can be subtle, they could be considered when directly comparing national alien species inventories (Jackson *et al.,* 2017). Native status is derived from the definition and an evaluation of the available evidence. Such evidence might be direct, such as, from fossil remains, specimens and first-hand accounts. However, native status is often evaluated indirectly from an assessment of the habitat, distribution, evolutionary history and life history of a species (Essl *et al.,* 2018; Hoagstrom *et al.,* 2009). For certain taxonomic groups evidence is particularly elusive. For example, rare soft-bodied organisms in deep marine habitats are rarely surveyed, and often given the status of "cryptogenic species" (Carlton, 1996; **Glossary**; **Chapter 2** for more examples). Assessments of native status are often made in plant and amimal surveys and are published in taxonomic checklists. In most cases the categorization is uncontroversial. However, in some cases the designation can have political and practical consequences.

6.6.1.2 Uncertainty of information on introduction pathways

Pathways of introduction, particularly in the marine realm, are not always known with high certainty (**Chapter 2**, **section 2.1.2**). Only occasionally are there documented deliberate releases, or clear evidence linking donor and invaded regions, and where species' life history and historical records point to an obvious introduction pathway. In most cases, vectors and pathways are assumed based on the biological and ecological traits of the species, the habitats they occupy in the native and introduced range, and the timing of first record, trade patterns, human use and vector activity (Faulkner *et al.,* 2016; Galil *et al.,* 2014; Hewitt *et al.,* 2004; Wonham & Carlton, 2005). For most species the precise details of their introduction history will not be known with any certainty (Wonham & Carlton, 2005). This might be the reason why only 10 per cent of the studies related to future scenarios and modelling of invasive alien species included pathways (**Chapter 1**, **section 1.6.7.3**). This limits options for governance because without adequate information on introduction pathways, it is difficult to implement policy for biosecurity, prioritize where to invest in interventions to manage biological invasions, or assign responsibility to actors responsible for unwanted introductions (**Chapter 2**, **section 2.1.2** for more information on knowledge and data gaps on introduction pathways and **Chapter 5**, **section 5.3.1** on pathway management strategies).

6.6.1.3 Balancing basic and applied research on biological invasions

There is no clear dividing line between so-called, "pure" research conducted solely for increasing information and applied research that has clear practical applications. Nevertheless, the distinction is made below to help us evaluate the balance of funding and resources devoted to different aspects of science.

In response to the problem, the number of peer-reviewed publications on biological invasions has increased steadily (Vaz *et al.,* 2017). These research publications can be broadly divided into basic research focusing on the process, patterns and impacts of invasive alien species, and applied research, focusing on their management and mitigation. While basic research allows us to understand temporal and spatial patterns of invasive alien species and their underlying mechanisms, applied research builds on the information generated from basic research to develop contextualized management strategies at varying spatial and governance scales.

Basic research dominates peer-reviewed publications on biological invasions (Esler *et al.,* 2010). This disparity may be accounted for by the publication of much applied research in grey literature, such as governmental reports (Lowry *et al.,* 2013). There is also large variation in the use of research methods in basic research. For example, nearly half (46 per cent) of the studies that attempted to understand the fundamental process of biological invasions are field observational studies, while less than one-fifth (18 per cent) were field experimental studies (Lowry *et al.,* 2013).

Similarly, a meta-analysis of biological invasions research from Latin American and Caribbean countries, between 2006 and 2008 found that only 5 per cent of publications focussed on invasive alien species management (Pauchard *et al.,* 2011). Of 185 articles, 57 per cent focused on analysing only one species and 43 per cent on more than one species. Invasion patterns were analysed in 39 per cent of them, invasion mechanisms in 25 per cent, bibliographic invasive alien species reviews comprised 12 per cent, impacts were the focus of 19 per cent, and new invasive alien species were reported in 17 per cent (Pauchard *et al.,* 2011). Basic research focussed on invasive alien species listing, population dynamics, biotic factors that promote invasion and ecological relationships (facilitation, competition and mutualism). The applied research focused on restoration, eradication or control measures (Pauchard *et al.,* 2011 and references therein). Publications on aquatic and marine invasive alien species in South American countries cover six major basic research themes: biology/ecology (58 per cent); invasive alien species new records (20.5 per cent); aquaculture (3 per cent); range expansions; genetics; and general reviews of aquatic species with a remarkably low number (all below 3 per cent), although the proportion of applied research papers is not reported (Schwindt & Bortolus, 2017). Uruguay is an example of a country that has developed both basic and applied research on terrestrial and aquatic non-indigenous and invasive alien species in the last 15 years (Brazeiro *et al.,* 2021; Brugnoli & Laufer, 2018).

Despite the apparent greater investment in basic compared to applied research, knowledge of some basic science questions is still inadequate globally. For example, in an evaluation of country-level checklists of invasive alien species, these were found to suffer from one or more of 10 different error categories, mostly related to poor information or measurement errors (epistemic uncertainties; McGeoch *et al.,* 2012). Important errors include: species misidentified as alien due to taxonomic uncertainty; failure to recognize invasive alien species as a result of insufficient surveying; overestimation due to the coarse spatial resolution of alien species distribution maps or species listing; delays in the publication of data; poor data management that leads to data being unfindable; incorrect decisions to list a species as "alien" (**Glossary**) due to inadequate and ambiguous information on species' native range; incorrect decision of listing species as "invasive" due to limited information on their population dynamics and impacts, and lack of evidence-based standardized and universal criteria for designating a species as invasive (**Chapter 5**, **section 5.6.2.5**, **Table 5.12**).

While acknowledging that the errors could not be eliminated completely, (McGeoch *et al.,* 2012) suggested some measures to minimize errors associated with country-level checklists, including expanding investment in invasive alien species research and monitoring, improving findability and accessibility of invasive alien species data, improving the speed at which a correction can be applied to a list, and improving transparency and repeatability of invasive alien species listing methods, along with standardized uses of terms and concepts.

Information generated from basic research is translated to management and policy responses through applied research. Poor representation of applied research in peer-reviewed publications (Esler *et al.,* 2010), might have contributed to the continuous increase in the number of alien species across taxonomic groups and biogeographic regions (Seebens *et al.,* 2018). Additional investment of resources for applied research would generate information suitable for managers and policymakers to make decisions.

6.6.1.4 Socioecological research to support policy and management

The prevention and sustainable management of biological invasions depends on an effective integration of environmental, social and economic components in management strategies (D. L. Larson *et al.,* 2011). This implies that an understanding of the socio-economic dimensions of biological invasions is as important as the knowledge held in the fields of biology, taxonomy and other scientific specializations. In spite of the obviously strong human and social dimensions of the invasion process, impacts of invasive alien species and their management (Shackleton, Shackleton, *et al.,* 2019), a 2017 study found that more than 90 per cent of research publications on biological invasions since 1958 were related to ecology and environment, while only 3.2 per cent of the publications primarily addressed socioecological dimensions (Vaz *et al.,* 2017). Similarly, only 3 per cent of 364 research articles related to invasive alien species produced by South Africa's iconic Working for Water Programme between 1995 and 2017 addressed human dimensions associated with invasive alien species (Abrahams *et al.,* 2019).

These scenarios suggest an under-representation of socially relevant research in biological invasion science; expanding it to include social dimensions of invasive alien species through interdisciplinary and transdisciplinary approaches will help to generate socially relevant additional data and information (Abrahams *et al.,* 2019; Esler *et al.,* 2010; Shackleton *et al.,* 2017; **Chapter 4**, **Box 4.5** and **section 4.7.1**). These approaches can not only better inform current management and policy decisions but may also better predict future invasions in an era of global change (Kueffer *et al.,* 2014). Furthermore, a transdisciplinary approach linking ecological and social sciences to generate data and knowledge is also helpful in building trust between communities and resource managers while managing invasive alien species that carry social value (Beever *et al.,* 2019). Ultimately, integrating knowledge systems will be the most fruitful approach to addressing biological invasions, and this includes crosscutting work with the fields of epidemiology, health sciences, economics, political science, sociology, psychology, anthropology, history and others.

6.6.1.5 Knowledge of Indigenous Peoples and local communities[15]

It has long been recognized that Indigenous Peoples and local communities hold unique knowledge on biodiversity. They often inhabit remote, biodiverse landscapes from which they derive diverse resources. Their knowledge may not be documented but may be important to understand ecosystem processes and resource management. Indigenous and local knowledge has been recognized and accepted as relevant to the development and good quality of life of Indigenous Peoples (Sillitoe & Marzano, 2009; Williams & Hardison, 2013). Nevertheless, Indigenous Peoples and local communities have often been excluded from decision-making and would wish to take more control over their cultural and intellectual knowledge (Bolhassan *et al.,* 2014). Historically the power imbalance between the holders and potential users of traditional knowledge have meant that the benefits derived from this knowledge have not been shared equally. Mistrust and misunderstanding has often developed in both directions between academic science and Indigenous Peoples and local communities (Bohensky & Maru, 2011; Mulligan & Stoett, 2000).

Internationally, the need to ensure equitable distribution of the benefits of knowledge and genetic resources has been recognized in the Nagoya Protocol (Buck & Hamilton, 2011). Though the Nagoya Protocol does improve the situation, it

15. Data management report available at: https://doi.org/10.5281/zenodo.5760266

is an intergovernmental agreement, and its implementation varies with jurisdiction and does not necessarily include the needs, aspirations and wishes of Indigenous Peoples and local communities. Furthermore, it is far from clear how knowledge, resources and data on invasive alien species themselves should be treated under the Nagoya Protocol because the Protocol is concerned with the benefits of biodiversity and invasive alien species are largely detrimental. The origin of the knowledge and genetic resources can be obscure, and species used by Indigenous Peoples and local communities traditionally are often alien species (e.g., de Almeida *et al.,* 2010). In the case of biological control agents best practices have been drawn up for access and benefit sharing (Mason *et al.,* 2018; D. Smith *et al.,* 2018). However, little consideration of the interests of Indigenous Peoples and local communities is given in these best practices.

Knowledge of invasive alien species by Indigenous Peoples and local communities is vital for not only the community itself, but also for policymakers and practitioners for the purpose of implementing control and management options (Williams & Hardison, 2013). An analysis of the sources of invasive alien species knowledge showed that the majority of Indigenous Peoples and local communities obtain their knowledge from self-learning, observation and experimentation. Another large group mentioned a mix of both contemporary and traditional knowledge sources. A smaller percentage relied on scientific knowledge, showing that Indigenous and local knowledge plays a big role. This also shows how important it is to incorporate both Indigenous and local knowledge and contemporary science while informing policies (Bolhassan *et al.,* 2014).

Communication of information and Indigenous Peoples and local communities[15]

A diverse array of stakeholders and institutions can work together to ensure smooth and effective communication of invasive alien species information. This is not only relevant to Indigenous Peoples and local communities but also to governments, policymakers and to bridge the gap between research and implementation (Barnard & Waage, 2004; Piria *et al.,* 2017). An analysis done on organizations with effective communication on invasive alien species showed that central governments (39 per cent) often have the financial capacity and resources to effectively communicate on invasive alien species. Thanks to their proximity to Indigenous Peoples and local communities, local governments (36 per cent) are also in a position to effectively communicate on invasive alien species. Person-to-person communication (individually; 32 per cent) can be effective as well but often faces geographical limitations and language barriers, which could lead to misinformation (Wald *et al.,* 2019; Zeng *et al.,* 2021). Finally, there are cases of effective communication on invasive alien species through community-led organizations (22 per cent), international and non-governmental organizations (16 per cent).

Knowledge and information needs of Indigenous Peoples and local communities[15]

For an effective and holistic involvement of Indigenous Peoples and local communities and other stakeholders in the control of invasive alien species and management of biological invasions, knowledge dimensions and improvement are vital (IUCN, 2000; Shine, 2003) both in science and practice. Many Indigenous Peoples and local communities (43 per cent of the reviewed case studies) are seeking scientific knowledge, through training, reading and contacting governments and non-governmental organizations, on how to control and manage invasive alien species. Thirty per cent are seeking Indigenous and local knowledge while 17 per cent combined both Indigenous and local knowledge and scientific knowledge. In only 10 per cent of the reviewed case studies, Indigenous Peoples and local communities seek additional knowledge through self-learning. From these findings, it is important for all players to make their data and information available and useful (Groom *et al.,* 2017) to all the stakeholders. The pace towards meaningful participation of Indigenous Peoples and local communities into various sectors of management could be fast-tracked to fill knowledge needs.

Indigenous Peoples and local communities and scientific knowledge[15]

There is significant agreement between Indigenous and local knowledge and science on invasive alien species. There are also some significant divergences, which suggests that continued dialogue will be useful (Byrne *et al.,* 2020; Lopian, 2005) including on species identification, their impacts and pathways of spread. For example, many Indigenous Peoples and local communities recognize invasive plants or animals as foreign in their areas. They were however ready to try different ways to make these useful, for example as food for livestock or food for humans. In other cases, Indigenous Peoples and local communities did not recognize some species as alien, while science would classify them as invasive alien species. This will likely have ramifications for effective communication and control measures, and the involvement of Indigenous Peoples and local communities in decision-making on environmental conservation in different settings. Indigenous Peoples and local communities are using science to supplement and further build on their understandings of invasive alien species. Some report that they supplement the knowledge they acquired from observation and experimentation (self-learning), with science-based training they received.

6.6.1.6 Information needed for invasion scenarios and models[16]

Given the high socioecological relevance of invasive alien species, it is essential to understand how future trends and impacts can be mitigated. There is a strong need to develop scenario narratives, and subsequent quantitative analyses, that assess possible outcomes of various potential trends in alien species distribution, spread and impacts on the environment, economy and society. Here the integration of all information, whether from scientific hypothesis testing or Indigenous and local knowledge, can be vital to developing realistic qualitative baselines to inform subsequent models. Together with robust and relevant targets (e.g., comparable to the 1.5°C target in the climate change discourse), scenarios can underpin decision-making by providing examples of various opportunities and avenues to reach these targets and so inform policy nationally and internationally.

Scenarios and model literature on biological invasions reveals several information needs about policy, future research and action on biological invasions. Most (about 70 per cent) of the studies including both scenarios and models were based on alien species distributions, with only 30 per cent of studies focusing on other biodiversity variables. Species abundance or impacts of invasive alien species and life-history information (e.g., growth, survival) are largely absent from scenarios (19 per cent, 9 per cent and 6 per cent of the publications). The literature is dominated by exploratory scenarios, while target-seeking and policy scenarios are only marginally represented (6 and 7 per cent respectively). Studies using expert-based opinion are practically absent in the available scenarios and models literature (only 1 per cent of papers consider expert opinions). Publications including scenarios and models largely neglect anthropogenic drivers such as demographics, governance, values and technology (each represented in less than 2 per cent of the publications), as well as interactions among different environmental, socio-economic, or cultural drivers. Finally, most studies do not consider policy and management (4 per cent and 21 per cent respectively).

6.6.2 Options for strengthening the generation and flow of information relevant for policy and management

When facing information gaps during the development of governance and policy for biological invasions, or when planning for their management, understanding the type of information gap provides an effective guide for identifying the most appropriate decision-support tools **(Figure 6.18)**. Using a risk assessment framework, it is possible to identify information gaps on the likelihood of invasion outcomes and use this to construct knowledge response options **(Figure 6.18)**. This is also a mechanism for directly connecting specific management scenarios with scientific support tools, including those discussed in the following sections and elsewhere in the assessment (**Chapter 5**, **section 5.6.3.2**).

6.6.2.1 Citizen science as an option for generating information on invasive alien species

Ecological research has long benefited from the voluntary participation of the general public, with participating members often being referred to as citizen scientists (Dickinson *et al.,* 2010; **Chapter 1**, **Box 1.15**; **Chapter 5**, **section 5.4.3.2.a**). In recent decades, citizen science has emerged as an indispensable tool for generating complementary data and information relevant to addressing the problems of invasive alien species and other global environmental changes (Theobald *et al.,* 2015) by tapping the potential of technologies from websites to smartphones for recording biological and environmental data (August *et al.,* 2015). Citizen science approaches can cover larger geographic areas and collect data over a longer period of time than professional scientists alone with the investment of comparable resources (McKinley *et al.,* 2017) and has contributed substantially to monitoring of global biodiversity (Chandler *et al.,* 2017). Some citizen science initiatives have filled geographic gaps for particular taxa (e.g., eBird; Amano *et al.,* 2016; B. L. Sullivan *et al.,* 2014). The options for citizen science to help fill information gaps on invasive alien species are therefore promising (**Chapter 1**, **Box 1.15**).

Commonly recorded parameters in citizen science initiatives are species name, geographic coordinates, photographs, species abundance and habitat description (Johnson *et al.,* 2020). From these primary data, several essential biodiversity variables (EBVs) such as species distribution, population abundance, phenology, demographic traits, migratory behaviours and disturbance regimes have been derived (Chandler *et al.,* 2017). Citizen science has been successfully used in spatio-temporal distribution mapping of invasive alien species (Brown *et al.,* 2018; Johnson *et al.,* 2020; Mannino & Balistreri, 2018; Marchante *et al.,* 2017), prediction of species' suitable climatic niches (Johnson *et al.,* 2020; Tiago *et al.,* 2017), early detection (Giovos *et al.,* 2019; Hiller & Haelewaters, 2019; Palmer *et al.,* 2017; **Box 6.20**), and understanding animal behaviour and plant phenology (Johnson *et al.,* 2020). In addition, citizen science can be helpful in spotting elusive invasive alien species, such as *Python bivittatus* (Burmese python; Falk *et al.,* 2016). It increases public awareness and involvement in the management of invasive alien species while generating scientifically valid data at a very low cost (McKinley *et*

16. Data management report available at: https://doi.org/10.5281/zenodo.5706520

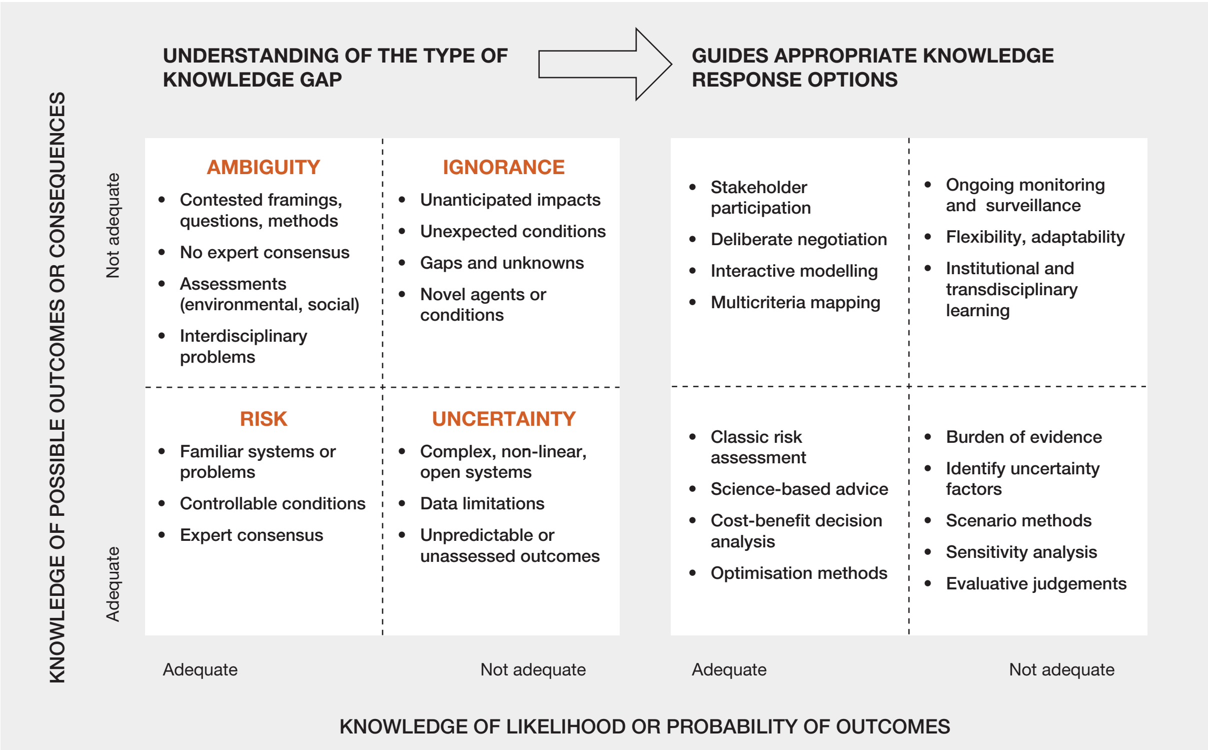

Figure 6 18 **Governing invasive alien species knowledge.**

An understanding of the multiple reasons for invasive alien species knowledge being either incomplete or inadequate (left), leading to the identification of appropriate options for strengthening the evidence base for invasive alien species governance (right; position of quadrants in right panel match those on left). The categories show how traditional, linear invasive alien species risk analysis methods (lower left quadrants in each panel) are on their own inadequate tools for governing the knowledge on invasive alien species needed to inform policy. For example, "Assessments" outline the broad suite of policy questions relevant to biological invasion, make it clear that no single solution is adequate and that options can be context dependent (Stirling, 2010). Adapted from Linke *et al.* (2016) https://doi.org/10.1007/978-3-319-27006-7_8, under license CC BY 4.0 and Stirling (2010) https://doi.org/10.1038/4681029a, under copyright 2010, Springer Nature Limited.

al., 2017; Palmer *et al.,* 2017). For recording species' distribution data, citizen science can be more cost-effective, nearly eight times less in case of Mosquito Alert, than the traditional expert-driven approach for data of comparable quality (Palmer *et al.,* 2017). Citizen science data can contribute substantially to existing species information data (e.g., early detection, species distribution range size, regional species pool) collected by professional scientists (Crall *et al.,* 2015; Palmer *et al.,* 2017; Soroye *et al.,* 2018).

Nevertheless, there are limitations to the use of citizen science, as shown by Pocock *et al.* (2019). There are large taxonomic and geographic gaps in data. For example, eight of the 26 citizen science initiatives with a web/mobile app evaluated focused on single (e.g., Mosquito Alert) or several priority invasive alien species (e.g., That's Invasive!, iMapInvasives) while the remaining initiatives (e.g., iNaturalist, eBird) include both native and alien species (Johnson *et al.,* 2020). Similarly, the number of invasive alien species focused citizen science initiatives leading to scientific publication was higher in Western Europe (11) and North America (10), and there was no such initiative in Asia (Johnson *et al.,* 2020). This is expected because 42 per cent and 32 per cent of all citizen science programme activities (N = 420) linked to biodiversity monitoring have been operating in North America and Europe, respectively (Chandler *et al.,* 2017). Expanding taxonomic and geographic coverage of citizen science initiatives increases the scientific values of the data generated. Adaptive sampling, whereby volunteers are guided to make observations in locations which will optimally improve species maps, has the potential to improve the effectiveness of citizen science for early warning of invasive alien species.

Citizen science programmes for invasive alien species detection and surveillance have recently expanded to marine systems (Delaney *et al.,* 2008; Thiel *et al.,* 2014), with active involvement of fishers, divers and the public at large. The contribution of citizen science is expected to expand over time, helping to address the limited funding and spatial/temporal coverage available with current programmes (Pocock *et al.,* 2018, 2019). However, some constraints could be considered in programme design and expectations, including selecting large-bodied, conspicuous taxa with easy-to-recognize diagnostic characteristics. In the future, genetic tools may be also adopted by citizen science programmes to enhance the potential taxonomic scope and for validation purposes (Ojaveer *et al.,* 2018).

Data from citizen science can contribute significantly towards a better understanding of biological invasions and other global environmental changes, provided that these data are adequately used in peer-reviewed publications (Theobald *et al.,* 2015). Some scientists are reluctant to use citizen science data, though relevant to their objectives, due to uncertainties related to data collection methods and data attributes (Burgess *et al.,* 2017). Accompanying citizen science data with metadata describing data quality, availability, conservation issues being addressed, study taxon and system, spatial and temporal scales of measurement, sampling intervals and data standardization protocols improves transparency and encourages scientists to use citizen science data (Burgess *et al.,* 2017).

In spite of the voluntary contribution of participants, citizen science data are not always openly shared (Groom *et al.,* 2017). In a recent study, nearly half (54 per cent) of the 26 invasive alien species-relevant citizen science initiatives evaluated did not share data with other similar initiatives or other biodiversity data-sharing facilities (Johnson *et al.,* 2020). Sharing data among other citizen science initiatives working in similar geographic regions/scales and taxa, and consolidating results in shared databases, would increase use values of citizen science data in scientific research, and policy and management decisions (Johnson *et al.,* 2020).

6.6.2.2 Professional networks and platforms for coordination and information exchange

It is important to understand patterns and processes of biological invasions at varying spatial scales for effective management. Several information systems and approaches are available at national, regional, continental and global

Box 6 20 **Citizen science for early detection and rapid response.**

After prevention, early detection and rapid response (EDRR) is the most effective and least costly way to manage invasions. The main hitch is the inability to generate enough resources to support a sufficiently large professional staff to survey with adequate frequency the vast amount of land and water that can house recently arrived invaders. Yet a well-informed, educated public can vastly increase the number of “eyes” on the lookout for incipient invasions; and individual citizens in the course of other activities have spotted hugely damaging invasive alien species in time to permit complete eradication before the species had spread widely. Such was the case of an individual sawing off an overhanging tree branch in Chicago and noting signs of *Anoplophora glabripennis* (Asian longhorned beetle; Kridel, 2008; Manier & Martin, 1998) and a recreational diver finding the famed *Caulerpa taxifolia* (killer algae) in a California lagoon (Muñoz, 2016).

Rather than simply relying on publicity about invasions and the hope that an alert citizen will happen to observe a recently arrived invader and know how to report it, several organizations have trained citizen volunteers and organized their search activities to maximize the probability of detecting recently arrived and potentially invasive plants. In the Australian state of Victoria, the Victorian Weed Spotters program, initiated in 2008, trains citizens to find and report state-prohibited weeds, and these reports are viewed as valuable components of the state programme to prevent weed establishment (Munakamwe *et al.,* 2018). In 2012 in the state of Washington, United States, the Pacific Northwest Invasive Plant Council organized an EDDR Citizen Science Invasive Plant Program to train volunteers to support county, state and federal management agencies to locate and eradicate invasive plants, a programme that has now expanded to the state of Oregon (PNW-IPC, 2018). Perhaps the most expansive such programme was organized by the Invasive Plants Atlas of New England (IPANE) in 2001 to integrate independent efforts of the six New England States. The programme, associated with an atlas of invasive plants in this region, trains volunteers both to find and to identify invasive plants, assigning particular monitoring routes. The programme is associated with EDDMapS (EDDMapS, 2019), a system of reporting and mapping alien species in the United States. However, the death in 2010 of the key innovator of the IPANE programme has led to a dearth of funds for training volunteers, and the programme itself has lapsed.

Aceves-Bueno *et al.* (2015) examined 83 citizen science programmes that entailed monitoring, of which five (including IPANE) targeted invasive alien species. In addition to substantial contributions to various resource management activities (including managing invasions), this study pointed to the important benefit of engaging a wide swathe of the public in recognizing and dealing with environmental issues, whether or not they associate themselves with formal programmes such as those described above. An important consideration, however, is the accuracy of monitoring records reported by citizen scientists, as noted by Crall *et al.* (2011) for an organized effort to monitor for invasive plants. This consideration supports the importance of substantial training for citizen volunteers, which bears a non-negligible cost. Another important step is to implement validations of citizen science invasive alien species data by experts (e.g., taxonomists) or through automated machine learning approaches (e.g., computer vision) or even by Artificial Intelligence.

scales to support information flow across jurisdictions (Katsanevakis & Roy, 2015; Mulligan & Stoett, 2000). Global analyses, for example, are essential for informing international policy to address the problems of invasive alien species, including those focussed on particular habitats and ecosystems. Collecting empirical data from diverse geographic areas will be improved by collaboration across strong networks of researchers, managers, practitioners and informaticians (Packer *et al.,* 2017). Previous and current examples of such collaborative networks include the Mountain Invasive Research Network (MIREN); European Information and Research Network on Aquatic Invasive Species (ERNAIS); Global Invader Impact Network (GIIN); Global Invasions Research Coordination Network; *Phragmites* Network (PhragNet); the Southern Hemisphere Network on Conifer Invasions (Packer *et al.,* 2017) and the Pacific Invasives Partnership, that is part of the Secretariat of the Pacific Regional Environment Programme (SPREP; for additional examples see **Supplementary material 6.3**).

International networks

GRIIS is supported by a broad collaboration of agencies and country experts. GRIIS provides open country checklists as well as a collated compendium of invasive alien species across countries (Pagad *et al.,* 2018, 2022). GRIIS data are openly available through an online repository, through the Global Biodiversity Information Facility (GBIF), and *via* country pages of the CBD Access and Benefit-sharing Clearing-House. GRIIS is founded on a transparent set of methods and biodiversity information standards (**section 6.6**) and provides both a baseline information source and mechanism for supporting an international information platform for invasive alien species (Pagad *et al.,* 2018, 2022).

At its 15th meeting, the CBD COP called for multiple international networks to continue supporting the implementation and monitoring of the Kunming-Montreal Global Biodiversity Framework (including Target 6 for invasive alien species), most notably the Statistical Commission, the Group on Earth Observations Biodiversity Observation Network (GEO BON), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem and Services (IPBES) and the Biodiversity Indicators Partnership (BIP) (CBD, 2022b), as well as the ISSG, GBIF and CABI (CBD, 2022c).

Previous and current international networks on invasive alien species that promote data sharing and collaboration include the ISSG, The Inter-American Biodiversity Information Network (IABIN), the Asia-Pacific Forest Invasive Species Network (APFISN), the European Network on Invasive Alien Species (NOBANIS) and CABI (which produces the Invasive Species Compendium). The European Alien Species Information Network (EASIN) provides opportunities for pan-European cooperation for sharing of information and assists implementation of European policies on biological invasions (Katsanevakis *et al.,* 2013). Also, a European Co-operation in Science and Technology (COST) action was launched to establish an alien species and citizen science network to develop and support citizen science initiatives (Roy *et al.,* 2018).

National and subnational networks

Examples of national or subnational initiatives, in this case from Europe are: a) a French working group on biological invasions in aquatic environments which aims at promoting expert knowledge, providing access to scientific information and guidance to decision-making for capacity-building to manage biological invasions (Sarat *et al.,* 2017) and b) a national code of conduct to prevent the introduction and spread of aquatic invasive plant species in the Netherlands (Verbrugge *et al.,* 2014).

Aquatic networks and information systems

An online information system on aquatic non-indigenous species and cryptogenic species (AquaNIS),[17] or species that might be considered to be invasive alien species, stores and disseminates information on invasive alien species introduction histories, recipient regions, taxonomy, biological traits, impacts and other relevant documented data (Olenin *et al.,* 2014). AquaNIS is being routinely updated by the supporting network (including by the members of the Working Group on Introductions and Transfers of Marine Organisms of the International Council for the Exploration of the Sea, ICES WGITMO) and contains information from various parts of the world.

Standard protocols

The Mountain Invasive Research Network was established in July 2005 during an international workshop on plant invasions into mountain regions in order to generate and share information of biological invasions in mountain regions of the world (Dietz *et al.,* 2006). The network has developed standardized protocols for data collection. Participating researchers use the same protocol while collecting data in mountain regions around the world. Use of empirical data collected from different parts of the mountain regions through this network has provided broad understanding of plant invasion patterns in mountain regions (e.g., J. M. Alexander *et al.,* 2011; Liedtke *et al.,* 2020; Seipel *et al.,* 2012). Similarly, the Global Invader Impact Network (GIIN) has developed a standard protocol for quantifying baseline ecological impacts (Barney *et al.,* 2015) and the methods have been already used to study impacts of species like *Impatiens glandulifera* (Himalayan balsam, Čuda *et al.,* 2017) and *Microstegium vimineum* (Nepalese browntop; Tekiela &

17. http://www.corpi.ku.lt/databases/index.php/aquanis

Barney, 2017). Some networks have become inactive and no longer collect data or update online resources.

Supporting active networks and platforms, re-activating previously established networks and developing new networks focusing on relatively less studied (e.g., wetlands) and difficult to quantify (e.g., marine) ecosystems creates opportunities for collecting and collating data using standardized protocols to improve knowledge of biological invasions.

6.6.2.3 Integration of information

Invasive alien species data in a biodiversity data context

The concept of essential biodiversity variables (EBVs) has been proposed to harmonize and unify efforts towards being able to provide regular, reliable and up-to-date information on key measurements of biodiversity change (Pereira *et al.,* 2013). Key elements of the EBV approach include aggregating and integrating biodiversity information (including on genes, species populations, traits, communities and ecosystems) across multiple sources, advancing the biodiversity information standards needed to achieve this, and state of the art modelling to deal with data gaps and provide estimates of uncertainty around projections. Generating such data products at a global scale has been challenged by the slow mobilization of data, inconsistent use of standards, a lack of standards and unevenness of data availability; problems which are now being overcome (Jetz *et al.,* 2019; Kissling *et al.,* 2015). Indeed, one of the goals of defining essential biodiversity variables (EBVs) is intended to be supporting the collections of up to date and higher quality raw observations of biodiversity **(Box 6.21)**. To create such essential biodiversity variables (EBVs) and tackle the underlying difficulties, automated workflows could make it feasible to repeat the process regularly and in a timely way (Best *et al.,* 2007; Kissling *et al.,* 2018). Workflows would output information that can be easily digested by policymakers and other stakeholders, who do not necessarily understand all the details of the workflow, but who need an appreciation of the data's limitations (Jetz *et al.,* 2019).

Integration of invasive alien species data through essential biodiversity variables

Three essential biodiversity variables (EBVs) have been identified as critical for measuring change in invasive alien species and underpinning invasive alien species indicators: alien species occurrence, the status of an individual or species as native or alien and invasive alien species impact (Latombe *et al.,* 2017; McGeoch & Jetz, 2019). To generate EBVs adaptable automated workflows are envisaged that can harvest raw data, aggregate them and standardize them to output the final EBVs product (Hardisty *et al.,* 2019).

Automated workflows for invasive alien species data integration and analysis have the advantage over bespoke programmes and semi-automated processes in that

Box 6 21 **Institutionalizing invasive alien species monitoring: a case study from India.**

Purpose

An example of how invasive alien species monitoring can be mainstreamed into government mandates is that of India's *National Tiger Estimation Program*. The government of India uses this program to estimate tiger populations at a national scale every fourth year and the program has been running since 2006. This monitoring not only produces an account of tiger numbers at a national scale, but also uses this charismatic species to garner resources and public support for protecting natural systems and their functions (Jhala *et al.,* 2021).

Approach

The National Tiger Estimation Program sampling protocol is developed to collect information on the distribution of important carnivore and herbivore animals, as well as their habitat quality. The protocol for assessing habitat quality is used for collecting information on invasive plants and native weeds. The primary objective of monitoring is tiger conservation, whereas weed monitoring is a conservation objective in its own right. The case demonstrates how, with appropriately well-integrated strategy and planning at national and sub-national scales, multiple biodiversity monitoring objectives can be simultaneously met – including invasion monitoring.

India has a large human resource with a mandate to monitor and protect forest ecosystems **(Figure 6.19)**. The National Tiger Estimation Program uses this trained capacity in collaboration with scientific institutions to sample forested areas at a 10×10 km resolution. Within every cell of 100 km^2, 20-40 plots of 10-30 m diameter are sampled to record the abundance of available plants. Since 2018, the data are recorded through an open-access mobile app (MSTrIPES) that stores this information in native language, along with geotagged photographs of the plot. These data are transferred to a cloud server, where existing algorithms compute trends in invasion of different species at a desired scale (e.g., Mungi *et al.,* 2020). The system can be further used to relate invasive alien species presence to herbivore and carnivore distribution. These analyses are reported back to the data collectors and managers, who can use them to prioritize evidence-based invasive alien species management and research (e.g., Mungi *et al.,* 2021).

Box 6 21

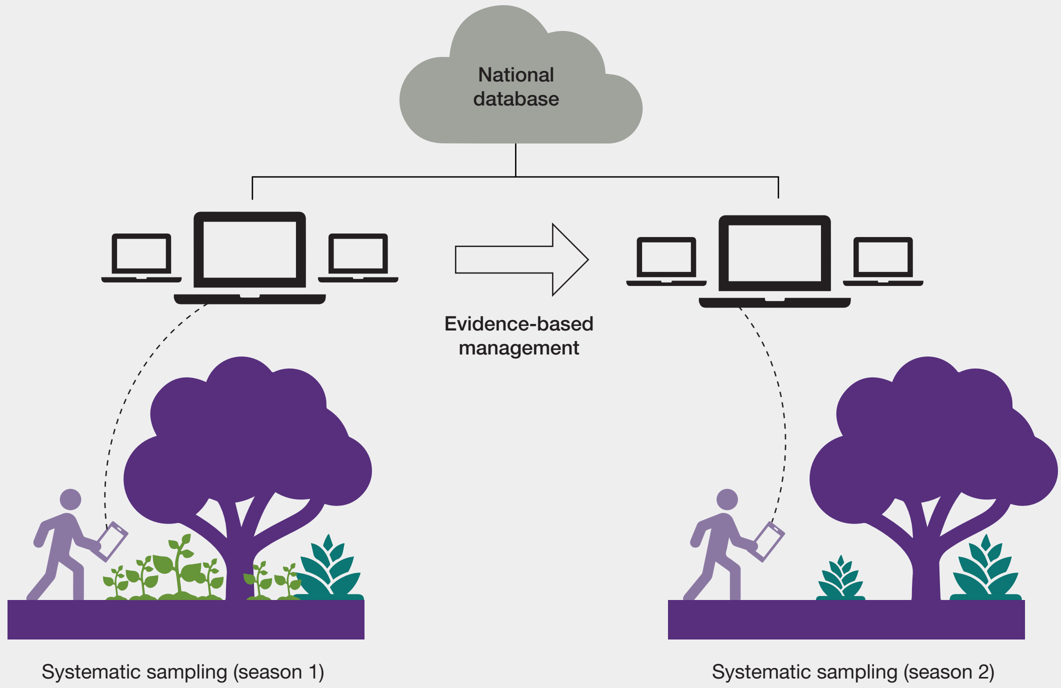

Figure 6 19 **Systematic monitoring scheme for invasive plants in Indian forests.**

The research team records abundance of invasive plants in sampling plots of every 100 km^2 cell across tropical India. The information is recorded on mobile apps that transfer field data to a regional workstation and cloud server, where spatial analyses are done to map and prioritize invasive plant management. The process is repeated at a national scale every four years. Recent monitoring in 2018 was the most extensive with more than 150,000 habitat plots sampled, revealing that 76 per cent of Indian forests invaded by different invasive plants at an alarming rate.

Combining monitoring and management

Importantly, the present monitoring protocol not only generates data, but it is also proactive. The sampling teams are equally involved in habitat management. In addition to evidence-based science, the present sampling protocol convinced the stakeholders to monitor invasive plants for conserving forest habitat that will help increase herbivore densities. This in turn will ensure sustenance of top predators, including the tiger. Serving as a unique platform to host scientific and management priorities, the present monitoring protocol is mandated by government agencies for continual application in India. This case can serve as an exemple for other countries who wish to mainstream and institutionalize invasive alien species monitoring with the limited resources availed for conserving iconic species.

they are flexible, repeatable, easily shared and can be repurposed (e.g., Seebens *et al.,* 2020; Reyserhove *et al.,* 2020). They can also take advantage of code written by many people and use shared resources. Workflows fit well into the ethos of open source and open science practice (Goble *et al.,* 2010). Although they are not yet widely used for invasive alien species data analysis, examples exist (De Giovanni *et al.,* 2016; Seebens *et al.,* 2020; **Box 6.21**).

A trial was conducted on the species distributions for three widespread invasive alien species to generate data ready to be used in essential biodiversity variables (EBVs; Hardisty *et al.,* 2019). This work identified some areas where research investment in data information systems for invasive alien species is needed. The workflow was based on the large open infrastructures GBIF and Atlas of Living Australia. They encountered several difficulties in fully automating their workflow, which served the purpose of identifying where further advances are needed in data integration methods and standards. Even though these infrastructures are based on the same standards, they encountered many inconsistencies between datasets, and required considerable manual expert input to complete the workflow. At a continental scale, a completely automated workflow has been created in the programming language R, specifically for generating indicators and models of invasive alien species distributions

(Oldoni *et al.,* 2020). Ultimately, it is conceivable that automated workflows will be built to generate essential biodiversity variables (EBVs) at a global scale; however, research and development are still needed to resolve data consistency problems and to develop analytic models.

6.6.2.4 Open science – open data for invasive alien species

The Open Knowledge Foundation defines “Open” as “… anyone can freely access, use, modify and share for any purpose (subject, at most, to requirements that preserve provenance and openness)” (Open Knowledge Foundation, 2021). The open data movement has been an important driving force increasing access to data (Molloy, 2011), and can do the same for invasive alien species data. This development has been motivated by the desire to improve transparency and reproducibility of science, but it also aims to improve the efficiency of science by avoiding unnecessary duplication of effort. The push toward open data is energized by the need for interoperability, particularly in complex, data-intensive science, such as invasion biology (Reichman *et al.,* 2011). Several widely supported declarations have been penned to encourage open data sharing in biology and science in general (Budapest Open Access Initiative, 2012; pro-iBiosphere Consortium, 2014). Indeed, there have been specific calls for data on invasive alien species, in particular, to be open (Groom *et al.,* 2015, 2017).

Related, but not synonymous with Open Data are the Findable, Accessible, Interoperable and Reusable (FAIR) data principles (Wilkinson *et al.,* 2016). Openness does facilitate findability, accessibility and reusability, which is why FAIR data often go hand-in-hand with open data. One motivation for promoting FAIR data is that research data are easily lost once the research they support has been published (Vines *et al.,* 2014). Without FAIR open data on invasive alien species, it is difficult to provide informed, integrated policy support nationally or globally.

Many online resources are available for sharing information about invasive alien species (**Chapter 5**, **Table 5.4** for examples). These resources vary considerably in how well they conform to the FAIR data principles, and how readily complementary information can be integrated from multiple sources to answer questions about biological invasions (**section 6.6.2.3**). There is considerable room for improvement and innovation to make resources for biological invasions more findable, accessible, interoperable and reusable.

6.6.2.5 CARE principles for Indigenous data governance

In analogy to the FAIR Data Principles, and complementary to them, are the Collective Benefit, Authority to Control, Responsibility and Ethics (CARE) Principles for Indigenous Data Governance [18] (RDA, 2019). These Principles try to address some of the historical and ongoing imbalances in governance of data concerning Indigenous Peoples. The letters of the acronym refer to Collective Benefit, Authority to Control, Responsibility and Ethics. The collective benefit, authority to control, responsibility and ethics (CARE) principles are not specific to data types and make no mention of particular issues related to data on biodiversity and invasive alien species. However, they are mentioned here because they are important guidelines concerning Indigenous and local knowledge and useful guidelines for the ethical management of any data needed on biological invasions.

6.6.2.6 Open access publication

Closed access to invasive alien species research has been recognized as a hindrance to conservation, wildlife management and policy on invasive alien species (Groom *et al.,* 2015; Jeschke *et al.,* 2019). This has led to the establishment of an international association, International Association for Open Knowledge on Invasive Alien Species (INVASIVESNET), to support the open dissemination of information on invasive alien species (Lucy *et al.,* 2016). Several open access academic journals have been established specifically on the subject of invasive alien species, their biology and management. For example, *Management of Biological Invasions*, *Aquatic Invasions* and *BioInvasions Records* are published by the Regional Euro-Asian Biological Invasions Centre (REABIC) and Neobiota is published by Pensoft. However, these discipline-specific journals publish only a small fraction of the academic research on invasive alien species. For scholarly literature in general, about 28 per cent is open access, but that percentage is increasing (Piwowar *et al.,* 2018). There is also clear evidence for an Open Access “advantage” in terms of citation (Eysenbach, 2006; Niyazov *et al.,* 2016; Piwowar *et al.,* 2018). However, this advantage may come at the expense of lower discoverability and access to other valuable research because closed access publications may be inaccessible to many researchers, particularly in low-income countries.

Information on invasive alien species is published in a wide variety of media and outlets, from journals and pamphlets to books. Much of this body of knowledge is not in publications dedicated to invasive alien species specifically but embedded within literature on biodiversity or ecology in general. Rapid access to scientific publications is essential to inform practitioners about a species and even to identify it in the first place. Nevertheless, those publications that are confined to paper are available for sale only for a short time after publication and then are available only in specialist or local libraries. Access to digital repositories, such as Zenodo, is providing a place where grey literature

18. https://www.gida-global.org/care

can be deposited for long-term preservation and findability. However, there also needs to be a change to the publishing culture so that all publishers consider the long-term archival of their work.

Researchers use indexes, such as Google Scholar, to discover potentially useful publications, though the accessibility of those publications varies. Gold Open Access publications are completely accessible to users and their licensing usually makes them reusable. However, at the other extreme are closed access publications that require large sums to access. Researchers are often adept at avoiding such costs and piracy of closed access publications is rife (Timus & Babutsidze, 2016). Even academics with legitimate access to scientific publications appear to find it easier to access papers illegitimately (Bohannon, 2016). This shows a demand for this scientific knowledge globally and a problem with the marketplace for academic knowledge (Björk, 2017), with particularly serious implications for research areas where environmental, social and economic costs of delays in the dissemination of information are serious, such as invasion biology.

6.6.2.7 Data standards for invasive alien species data

Global standards for invasive alien species information facilitate rapid, unambiguous communication and enable the delivery of indicators of invasions, regional comparisons, which in turn feed into policy support tools. Standards for data exchange of taxon observations have been around for a number of years and Darwin Core is predominant among them. Darwin Core has been adopted by GBIF (Canhos *et al.,* 2004). Until recently Darwin Core lacked some important features to make it useful for communicating about invasive alien species. However, recently proposals have been made to include the degree of establishment and introduction pathway within Darwin Core, together with controlled vocabularies for those terms. These changes have now been implemented by GBIF and it now requires data publishers to embrace these terms and use them (Groom *et al.,* 2019).

Few other official standards exist for specific data related to invasive alien species. However, several quasi standards exist under the umbrella term "framework". For example, Hulme (2009) published a framework for introduction pathways. The intention was to have a globally applicable pathway classification that could be used for all invasive alien species, whatever their natural habitat. This was to address a need to monitor pathways of introduction and to communicate pathway information in a more comparable way. A guide has been written to help users of the framework to interpret different pathway categories and improve consistency (IUCN, 2017). The CBD (2014a, 2014b) has developed a pathways framework, as has the ISSG (Pagad *et al.,* 2015). This is tied to the Aichi Biodiversity Target of identifying, prioritizing and managing pathways of invasive alien species. Armed with such information it becomes easy to provide the evidence to support policy on pathways (**Chapter 5**, **section 5.3.1**).

Nevertheless, the framework is still only a standard suitable for human interpretation and unsuitable for machine interoperability. Only by formalizing the framework as a data standard can the latter be achieved. The Invasive Organism Information Task Group of the Biodiversity Information Standards organization has proposed changes to Darwin Core to incorporate pathway vocabulary, adopted by the CBD (Groom *et al.,* 2019). These recommendations have been ratified by the Biodiversity Information Standards organization who manage Darwin Core but may take a number of years to be adopted by the wide range of stakeholders who gather, manage and use these data. Progress towards machine readable standard data will make the vision of creating rapid and reliable workflows towards policy-relevant indicators feasible (McQuilton *et al.,* 2016; Rocca-Serra *et al.,* 2016).

6.6.3 Tracking and reporting on policy and management effectiveness: indicators, metrics and datasets to support policy

Reporting on the effectiveness of policy in generating progress towards targets and goals for invasive alien species takes place at multiple jurisdictional levels – from global to subnational. Regardless of the level at which evaluation of policy effectiveness is needed, such evaluation relies on relevant and adequate data and analysis. The information needed for reporting on invasive alien species includes (i) the identity and spread of invasive alien species, including the pathways *via* which this occurs, (ii) the type and severity of impacts incurred by particular invasive alien species, (iii) societal values impacted, including for example, biodiversity, agricultural and human and animal health and (iv) data on management interventions (left of **Figure 6.20**; **Box 6.22**).

A key instrument to progressing efforts to deal with invasive alien species is the use of a range of indicators of status and trends in invasive alien species. These indicators are designed to be used for reporting on policy goals and targets at national and global scales, including the Kunming-Montreal Global Biodiversity Framework (Target 6 for invasive alien species) and the Sustainable Development Goals. Importantly, such indicators, supported by relevant metrics and data, have a longevity beyond medium term reporting cycles so that progress can be tracked consistently over the long-term. To date, invasion indicators that are global in scope have been used across five Global Biodiversity Outlook Reports (SCBD 2001-2020), and to

Box 6 22 Sustainable delivery of information on invasive alien species for reporting on policy and management effectiveness.

Figure 6.20 below shows a proposed framework for closely linking invasion targets with the data and tools needed to measure and make progress to achieving them (McGeoch & Jetz, 2019). Combined in digital, modular platforms with custom tools and interfaces the framework enables both evaluation of global progress and decision-support for local actions.

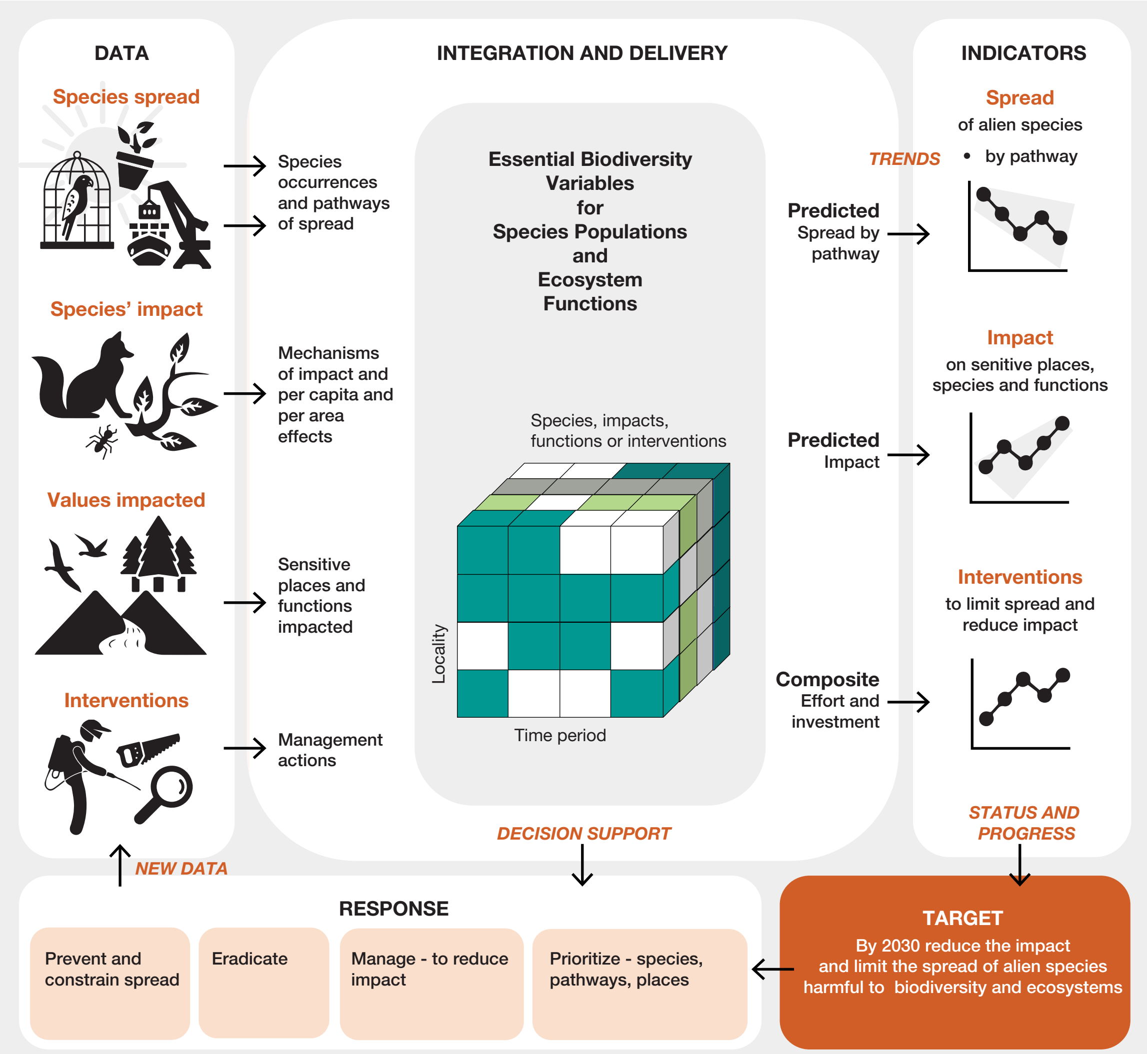

Figure 6 20 **Proposed framework for closely linking invasions targets with the data and tools needed to measure and respond to them.**

The target (lower right) frames and guides data generation, integration and delivery *via* modelled decision support products and indicators, to target responses for more effective intervention and a next generation of improved outcomes on biological invasion. Data (left) on the three key dimensions of the problem, (1) spread, (2) impact (both its type and consequence) and (3) interventions are integrated in a set of workflows that combine primary evidence in informatics infrastructure. Data providers are multiple and include for example GBIF, GRIIS, CABI – Invasive Species Compendium, the IUCN and the World Database on Protected Areas (WDPA). Currently data on interventions for invasion and their effectiveness are poorly collated with no dedicated infrastructure. Integration and delivery (centre) are at the core of the framework. Data, with the support of models, are used to predict occurrences or abundances of invasive alien species across pathways of introduction and spread (including establishment) and over contiguous spatio-temporal units, representing the Species Populations Essential Biodiversity Variables (EBVs) for invasive alien species (GEO BON). This is supplemented with data and essential variables that capture ecosystem functions and sensitive priority areas impacted by invasion, as well as data on management actions to predict impact and quantify intervention effort. Indicators (right) build on the delivered invasion-relevant Essential Biodiversity Variables and are

Box 6 22

Figure 6 20

populated using modelled predictions with associated uncertainty on spread, impact and interventions (Vicente *et al.,* 2022). Response (lower left) consists of the four major interventions – prevention, eradication, management and prioritization. These activities are guided by decision-support tools and products (such as alien species distribution maps for protected areas and location-specific automated alerts for new invasions) provided from integration and delivery directly or *via* indicators. The responses in turn deliver much-needed new data, including data on intervention effort and success. From McGeoch & Jetz, (2019), https://doi.org/10.1016/j.oneear.2019.10.003, under Copyright Elsevier.

support the summary for policymakers of the IPBES Global Assessment (IPBES, 2019c); and have been considered to various degrees in national reporting (Secretariat of the CBD, 2020; J. R. U. Wilson *et al.,* 2018). Proposed headline indicators for monitoring the implementation of the Kunming-Montreal Global Biodiversity Framework have been published by the CBD (2022b), including a specific headline indicator for biological invasions "6.1 Rate of invasive alien species establishment".

Indicators of biological invasion are broadly classified into indicators of (1) the drivers that facilitate biological invasion, (2) the size of the invasion problem (pressure indicators), (3) impacts on biodiversity and society (state indicators) and (4) societal responses to invasion (response indicators; Butchart *et al.,* 2010; McGeoch *et al.,* 2010a; **Table 6.2**). Under a theory of change framework, response indicators are now further subdivided (into input process, output, outcome, impact) to capture the stages of implementation necessary to bring about the desired progress (OECD, 2019; J. R. U. Wilson *et al.,* 2018; **Table 6.11**).

Although central to tracking the success of interventions to prevent and reduce the harm caused by invasive alien species, the development, adoption and fitness for purpose of invasion indicators has to date been inadequate (Vicente *et al.,* 2022). No existing indicators meet all the criteria ideal for robust, policy-relevant indicators (Vicente *et al.,* 2022). Challenges also include invasive alien species indicators that are not supported by robust and repeatable scientific methods, a lack of indicators to report on some important aspects of the problem, and indicators that are reliant on increasingly old data, which as a result are not able to report on recent progress (Vicente *et al.,* 2022). Although a number of intergovernmental and research partnerships have supported this endeavour over the last decade, including GBIF, IUCN Species Survival Commission (SSC) ISSG and GEO BON, two key factors are responsible for the slow progress. First, there has not yet to date been widespread agreement and adoption of a coherent, fit for purpose suite of indicators that can be used for long-term reporting **(Table 6.2)**. Second, there has to date been no institutional home with the resources and capacity to drive the research and reporting needed to sustain a robust suite of invasive alien species indicators (Vicente *et al.,* 2022). One of the evident outcomes of this is that the multiple indicators identified in CBD-related documentation change from reporting period to reporting period, and some are not able to be delivered or updated at the end of reporting cycles.

The options for strengthening the information value of invasive alien species indicators and their relevance for policy are clear. These include:

- Invest in the on-ground monitoring systems needed to deliver up to date information on the identity, spread and impacts of invasive alien species; and on the implementation and effectiveness of responses, including the implementation and effectiveness of management actions (Latombe *et al.,* 2017);
- Complete the research needed to support robust scientific formulations of indicators, the metrics on which they based, how they are modelled and interpretation of the uncertainty associated with them (Jetz *et al.,* 2019; McGeoch & Jetz, 2019);
- Establish a stable partnership to support invasion indicators that has the scientific expertise, data and analytic capacity and resourcing necessary to sustain these indicators over the long-term;
- Support the open infrastructures, data sources and collation processes required to aggregate and inform invasive alien species indicators, such as GBIF and the Global Register for Introduced and Invasive Species that jointly provide the data foundational to informing on invasive alien species (Pagad *et al.,* 2018);
- Assess and progress the extent to which each indicator can be downscaled and expressed at country level and the extent to which they are suitable for use at a national scale (J. R. U. Wilson *et al.,* 2018).

Table 6 11 **Categories of currently used invasive alien species indicators at a global scale, their information content and development needs.**

The indicators listed are phrased broadly to represent multiple closely related indicators that have been expressed in slightly different ways across the literature, policy documentation and historical, current and proposed reporting cycles.

Indicator category	Indicator expressed in an inclusive general form, encompassing relevant alternative formulations of closely related indicators	Data sources	Development needs
Driver	Trends in pathways of introduction and spread	No current FAIR source	Although raw trends can be produced, research is needed to develop these into a robust indicator with estimates of uncertainty (McGrannachan *et al.,* 2021) GRIIS has the potential to inform this indicator in future
Pressure	Trends in numbers and spatial distribution of invasive alien species and their impacts	GRIIS GBIF First Records	Further research to deliver downscaling to countries
State	Trends, mechanisms and severity of invasive alien species impacts	Environmental Impact Classification for Alien Taxa (EICAT) GRIIS GBIF	IUCN Red list Index for invasive alien species is well established EICAT progressing but still under development Downscaling to ensure relevance to countries required
Response and Theory of Change sub-categories (section 6.2.1)			
Input	Trends in the allocation of resources towards the prevention or control of invasive alien species	IUCN SSC ISSG	The methodology could be improved through peer review and further development
Process	Trends in establishment and national adoption of international agreements relevant to the prevention and control of invasive alien species	IUCN SSC ISSG	Reaching saturation as the majority of countries adopt most agreements, but still room for improvement on the most recently adopted (2010) relevant convention (BWM Convention)
	Trends in numbers of countries with national legislation and other policy measures relevant to the prevention and/or control of invasive alien species	IUCN SSC ISSG	Further research and development are needed to assess cross-country comparability of policy instruments and their fit for this purpose
Output	Trends in the prevention, eradication and control of invasive alien species	No current FAIR data source Data largely not been collected and collated by countries	Disaggregation is needed for priority sites
	Growth in information relevant to informing policy on invasive alien species prevention and control	GBIF First Records GRIIS EICAT	There is potential for disaggregation from a global indicator of information status on species populations
Outcome	Trends in successful eradications	Database of Islands and Invasive Species Eradications (DIISE)	Currently limited to birds and mammals on islands Requires taxonomic and geographic expansion
Impact	Improvement in conservation status of species threatened by invasive alien species	IUCN Red List of Threatened Species	Requires expression at sub-global scales

6.7 TRANSFORMATIVE OPTIONS FOR ADDRESSING THE PROBLEM OF BIOLOGICAL INVASIONS

This section addresses the following question: What will it take to tip the current systems – including socio-institutional, socio-technical and socioecological systems – that drive and manage biological invasions in the direction of sustainability (Loorbach *et al.,* 2017; Westley *et al.,* 2011)? The section begins by describing what integrated governance for biological invasions is and why such an approach is relevant for biological invasions. It also suggests how this approach could be implemented using a set of strategic actions and governance system properties that will bring about transformative change, challenges to setting these actions in motion and how these could be overcome.

Sustainability science has emerged as an applied field in response to the need for sustainable development and acknowledging the complexity of the socioecological systems that need to be governed and managed in order to achieve it (Clark & Harley, 2020; Loorbach *et al.,* 2017). The problem of invasive alien species is one instance of a threat posed to both society and the environment because of unsustainable development. Invasive alien species are a direct driver of nature's decline (IPBES, 2019c), and tackling these is therefore key to bending the curve of biodiversity loss. Therefore, invasive alien species as a problem share many of the features of sustainable development challenges, and an awareness of the risks posed by these is essential to the effective delivery of several of the SDGs (in particular, goals addressing the conservation of marine biodiversity (Goal 14) and terrestrial biodiversity (Goal 15, including but not restricted to target 15.8), food security (SDG 2), sustainable economic growth (SDG 8), sustainable cities (SDG 11), as well as climate change (Goal 13) and health and wellbeing (Goal 3)).

Many of the options for achieving goals and targets for invasive alien species will be enabled by systemic changes that parallel and reinforce the solutions needed to achieve sustainability more broadly (Chan *et al.,* 2020; S. Díaz *et al.,* 2019). Transformative change becomes necessary to achieve sustainable management of biological invasions because, like other key environmental threats, biological invasion is driven by demographic, social, economic and technological factors (Visseren-Hamakers *et al.,* 2021; **Chapter 3**, **sections 3.3** and **3.6**).

This assessment therefore builds on the sustainability science framing of the IPBES conceptual framework and enablers of transformative change (S. Díaz *et al.,* 2019; Scoones *et al.,* 2020): To reverse nature's decline while addressing inequality, a "fundamental, system-wide reorganization across technological, economic and social factors making sustainability the norm" (S. Díaz *et al.,* 2019). In this context, governance is the "formal and informal (public and private) rules, rulemaking systems and actor networks at all levels of human society that enable transformative change" (Visseren-Hamakers *et al.,* 2021). For invasive alien species, transformative change depends on a system-wide reorganization, including technological, normative, economic and social factors, needed to achieve the goals enshrined in multilateral agreements and national strategies.

6.7.1 Integrated governance can bring about transformative change that improves the management of biological invasions

To bring about such system-wide reorganization for managing biological invasions, the approach could focus on the "deeper system properties" (Leventon *et al.,* 2021; Meadows, 1999) that characterize biological invasion governance. The full suite of governance models, policy instruments and support tools and methods identified in this chapter **(Table 6.1)** are available as options which, in combination, can be drawn on to achieve this ambition. These include (*sensu* Scoones *et al.,* 2020):

1. structural options that involve fundamental changes to the way policy and management of biological invasions is organized, legislated, regulated (**sections 6.3** and **6.5**);

2. systemic options that involve "changes targeted at the interdependencies of specific institutions, technologies and constellations of actors across scales and geography to steer complex systems" of stakeholders and Indigenous Peoples and local communities contributing to, influencing and affected by invasive alien species (**sections 6.2** and **6.6**); and

3. enabling options that "foster the human agency, values and capacities necessary to manage uncertainty, act collectively, identify and enact pathways" to futures where the risks and negative impacts of invasive alien species are substantially reduced (**sections 6.2** and **6.4**).

As defined in **Box 6.5**, integrated governance for biological invasions means establishing relationships between the roles of actors, institutions and instruments and involving, as appropriate (in other words the specific features will depend on the national and local contexts), all those elements of the socioecological system that characterize biological invasion and its management, for the purpose of identifying the strategic interventions needed to improve prevention and

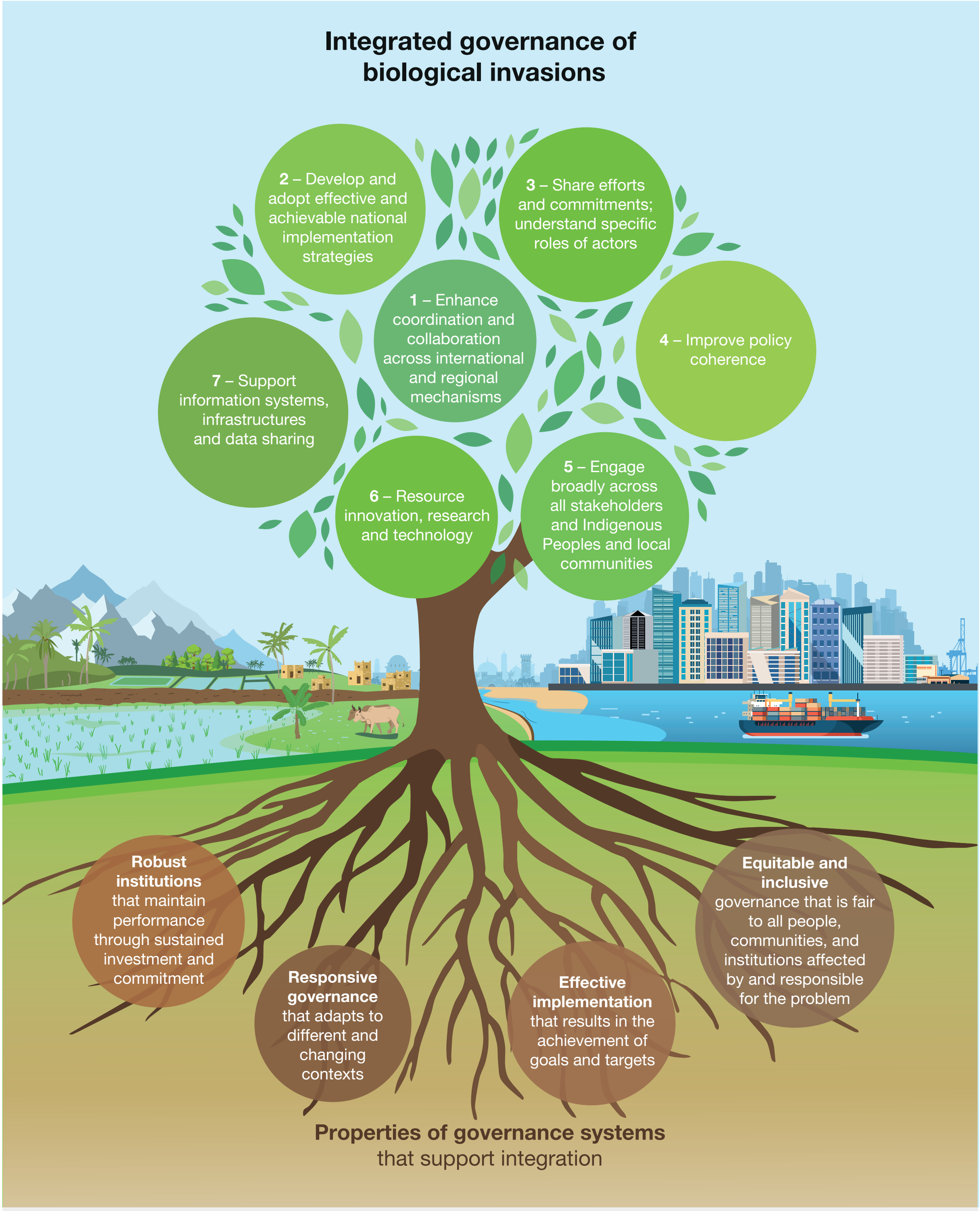

Figure 6 21 **Integrated governance for biological invasions.**

A context-specific integrated governance approach of biological invasions is enabled by a governance system with properties that support integration and a set of strategic actions that together are designed to bring about the progress needed to meet national and international goals and targets for biological invasions. Integrated governance is rooted (below) in four essential properties of governance systems that support the strategic actions (above) to be achieved. Together, the properties and actions will bring about the step change needed for effective and sustainable management of biological invasions. Integrated governance for biological invasions reinforces the enabling conditions identified as necessary to fulfil the 2030 mission of the Kunming-Montreal Global Biodiversity Framework. An integrated governance approach activates specific strategic actions that promote transformative change to meet the goals of preventing and controlling biological invasions.

Figure 6 21

The strategic actions (branches) are:

1. Enhance coordination and collaboration across international and regional mechanisms.
2. Develop and adopt effective and achievable national implementation strategies.
3. Share efforts and commitments, and understanding of the specific roles of all actors.
4. Improve policy coherence.
5. Engage broadly across governmental sectors, industry, the scientific community, Indigenous Peoples and local communities and the wider public.
6. Support, fund and mobilize resources for innovation, research and environmentally sound technology.
7. Support information systems, infrastructures and data sharing.

The proposed strategic actions are enabled when the system-wide properties of governance (roots) are robust, equitable and inclusive, responsive and focused on effective implementation. The numbers on the branches do not imply a ranking.

control outcomes **(Figure 6.21)**. While at face value this appears to be a monumental task, many of the processes and elements for preventing and controlling invasive alien species are already established and in play. Enhancing, strengthening and improving implementation and better integrating the actions and system properties that make up integrated governance for biological invasions could bring about a step change in progress.

Integrated governance for biological invasions is also establishing relationships between the roles of actors, institutions and instruments to ensure a shared, connected, coherent and differentiated effort to manage biological invasions **(Figure 6.23)**. It also involves the engagement of all the appropriate elements of the socioecological system that characterize biological invasion and their management to define the best strategies in those areas when invasive alien species impose socio-economic impacts (Bacher *et al.,* 2018). Last, it acknowledges that good governance, while essential to achieving sustainable outcomes for the prevention and control of invasive alien species, is somewhat of an experiment (Clark & Harley, 2020) and would therefore need to be adaptive as well as coordinated to facilitate learning (Brauman *et al.,* 2020). **Figure 6.21** illustrates "integrated governance for biological invasions" as the framework by which transformative governance (**Glossary**) for invasive alien species could be achieved, encompassing seven strategic actions and four governance system properties.

6.7.2 Strategic actions

6.7.2.1 Enhancing coordination and collaboration across international and regional mechanisms

The most important proactive (e.g., border control) and reactive (e.g., eradication) measures to address biological invasions are administered at the national or subnational level. However, the nature of biological invasions means that multilateral and transnational approaches are also needed. While there is no shortage of organizations focused on addressing the problem of biological invasions, one of their main limitations has been their disconnected nature. Cooperation amongst different regional and national efforts will not arise spontaneously and will need concerted leadership, resourcing and commitment from governments and institutions at the highest level (Leclère *et al.,* 2020; Ruckelshaus *et al.,* 2020; **section 6.2.3.1**). Therefore, establishing or enhancing global coordination mechanisms (similar to the Convention on the Conservation of Migratory Species of Wild Animals (CMS)) for biological invasions, or embedding this role into existing coordinating bodies (e.g., the CBD), are options for achieving one of the key strategic actions for transformative progress **(Figure 6.21)**. Such coordination mechanism could promote the exchange of best practices and other knowledge between regions and nations, help to establish the appropriate roles and responsibilities of actors (Stoett, 2007), enable global species listings and strengthen the effectiveness of the Inter-Agency Liaison Group on Invasive Alien Species. While the CBD currently covers invasive alien species as a cross-cutting issue, this assessment has amassed sufficient evidence to suggest the theme needs a more pronounced coordinating mechanism at the global level.

6.7.2.2 Developing and adopting effective and achievable national implementation strategies

Failure to adopt existing guiding principles (e.g., **Table 6.3**), and to implement legislation and action plans, have been a central impediment to progress on invasive alien species targets (**sections 6.2.2** and **6.3.1.4**). Implementation focused strategies for biological invasion management can assist in overcoming this hurdle **(Figure 6.21)**. This can be achieved by revisiting implementation strategies at all levels of governance, and in particular at those most relevant to the strategic actions identified in **Figure 6.21**. Options for this include, amongst others, consistent enforcement of relevant law (Chan *et al.,* 2020) and investment in monitoring and learning from the successes and failures of interventions **(Box 6.23)**.

Feeding this information into response-focused theory of change indicators, including indicators that track the allocation of resource inputs, the establishment and

Box 6 23 Overcoming the implementation gap for invasive alien species.

Two key hurdles to improving the management of invasive alien species, are:

1. the need for more effective prioritization of where and when to intervene, and
2. the lack of information on which interventions are most successful and in which contexts.

Both these hurdles can be overcome by generating essential data and knowledge on resource inputs, processes, outputs and outcomes (OECD, 2019) of efforts to manage invasive alien species **(Box 6.6; Figure 6.22)**.

Resource inputs: How much are governments and other responsible actors spending on invasive alien species management? What are the gaps in appropriately qualified, existing capacity and expertise that can be filled?

Processes: What coordination and oversight mechanisms are in place, from local communities to governments, to enable investment and ensure effective implementation of invasive alien species management?

Outputs and actions: What new or strengthened instruments are in place to improve policy coherence, to guide strategic investment and to adequately share and differentiate responsibilities for invasive alien species prevention and control?

Outcomes: Has the rate of new introductions and newly established invasive alien species declined? Has their spread and impact been reduced?

A sustained information platform for invasive alien species can then deliver this information when it is needed and to where it is needed (**section 6.6.2.4**). With the data that are collated, instrumental indicators will be able to report on progress and, based on the knowledge they provide, to iteratively refine and improve the efficiency of responses (i.e., adaptive governance, planning and management; **Figure 6.22**). Together, this approach can catalyse the collection of the information most needed to manage invasive alien species, reduce uncertainty and improve efficiency in decision-making. It can also support reporting on sustainable development and progress to meeting national and multilateral goals and targets for invasive alien species. Clearly linked to and embedded in national strategies for invasive alien species, this approach can leverage the responses needed to overcome current implementation gaps and provide a backbone for integrated governance.

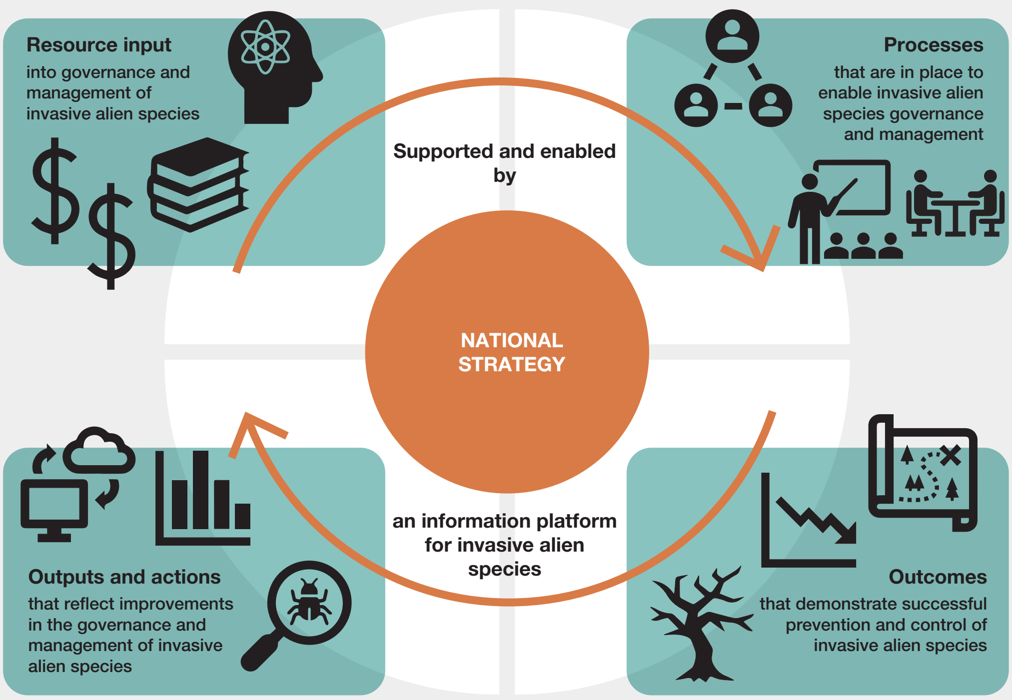

Figure 6 22 Monitoring progress in four types of responses to invasive alien species to leverage activity to overcome the implementation challenge.

If these responses are effective, they should manifest in a reduction in the numbers, spread, impacts and costs of invasive alien species.

uptake of implementation processes, and the outputs and outcomes of these interventions are in line with the Kunming-Montreal Global Biodiversity Framework (CBD, 2022a; OECD, 2019; J. R. U. Wilson *et al.,* 2018; **Table 6.11**; **Box 6.23**). While a strategy is necessary for effective governance at multiple levels and in multiple sectors (**section 6.3**), national scale strategy can be particularly instrumental in achieving the scope and cohesion needed to implement action both above (multilateral) and below (at local and subnational) national government.

6.7.2.3 Sharing efforts and commitment, and understanding the specific role of all actors across governments, Indigenous Peoples and local communities, and industries

The principle of shared, connected and specific roles builds on the fact that individuals, communities, industry and governments share the benefits from nature and therefore also share the responsibility of mitigating the risks imposed by drivers of change such as invasive alien species. This definition is a contextual application of the international environmental law principle of common but differentiated responsibilities (Stone, 2004) as it considers all parties as equally responsible for addressing the problem, but their knowledge and tasks are clearly differentiated based on their relationship to the problem; this is fundamental for a successful application of integrated governance for biological invasions **(Figure 6.23)**.

The management of biological invasions is a collective effort where individuals, communities, industry and governmental agencies play a unique but coordinated role. Such coordination builds from engagement of all actors concerned with the mitigation needed to avoid environmental, economic and health impacts from invasive alien species **(Figure 6.23)**. This can be achieved through a co-production approach that acknowledges that all

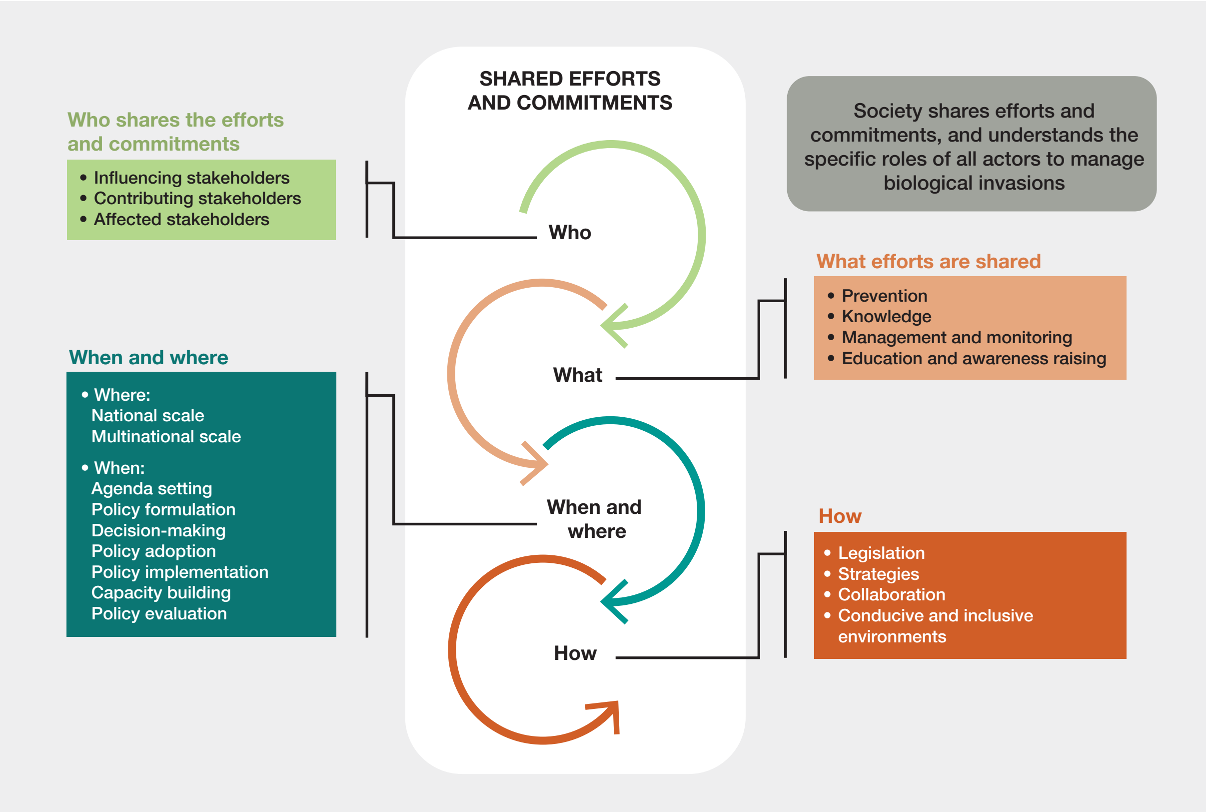

Figure 6 23 **Sharing efforts and commitments, and understanding the specific role of all actors for the prevention and control of invasive alien species.**

This approach emphasizes that people, knowledge, governance and policy instruments have a specific role and need to be connected. The shared efforts and commitments principle means that all actors are part of the solution, so that actions to manage invasive alien species are connected (i.e., there is effective flow of communication between all actors) between people, knowledge, governance approaches and policy instruments. The understanding of the specific roles of all actors, is an idea based on the principle that every institution and stakeholder have a specific part to play in the coordinated and coherent strategy that is the integrated governance for biological invasions.

stakeholders involved hold relevant knowledge and expertise; and that defines strategic connections between people, knowledge, governance approaches and policy instruments (Lemos *et al.,* 2018; Turnhout *et al.,* 2020).

Concrete actions by industries involved in potential pathways for invasive alien species are necessary to increase compliance with current legislation, and voluntary codes of practice can limit the biosecurity risks posed by industrial actors. International organizations can help here: for example, tourism can operate as a pathway and the World Tourism Organization (UNWTO) was asked at CBD COP15 to examine collaborative efforts to reduce invasive alien species introductions. The engagement of the general public *via* citizen science platforms, awareness campaigns, or community-driven eradication campaigns is critical for generating shared efforts and commitments by understanding the specific role of all actors play for addressing the invasive alien species problem. Also critical is the context-contingent involvement of specific stakeholders (e.g., agriculture producers, hobby associations, leisure groups) and Indigenous Peoples and local communities. The engagement and empowering of Indigenous Peoples and local communities is a crucial part of developing inclusive systems that recognize the rights of these communities and their knowledge, practices and values in the management of biological invasions. Such engagement strategies can generate ownership of biological invasion management while supplementing surveillance and management efforts.

6.7.2.4 Improving policy coherence

Global environmental change (with climate change as an example)

Invasive alien species impacts can compound the negative effects of climate change on good quality of life, acknowledging that this outcome is dependent on the species, regions and local conditions involved (e.g., Bradley & Wilcove, 2009; Shabani *et al.,* 2020). It is important that the transformative change needed to prevent and control invasive alien species is not neglected in the current context of a necessarily strong policy focus on climate change. The direct effects of invasive alien species on biodiversity are one of the ways in which the consequences of climate change are translated into direct negative outcomes for nature's contributions to people and good quality of life. Although invasive alien species and climate change are projected to affect fewer species than land-use change, these drivers can interact to become critically important at local scales and can impact people directly (Leclère *et al.,* 2020). The integration of invasive alien species and climate change policy considerations through environmental governance more broadly are options supported by a groundswell in forward-looking thinking and strategy on how to achieve environmental sustainability. The integrated governance approach that focusses attention on the intersections, linkages and trade-offs – and the research, policy and governance instruments needed to achieve complex objectives – is an option for advancing this ambition (J. Liu, Dou, *et al.,* 2018). The exploration of governance arrangements across many different goals and drivers of change is an option for overcoming the multiple needs for building and maintaining reflexive (learning by self-reflection) governance capacity (Clark & Harley, 2020).

Coherence between sectorial policies and institutions

One of the main reasons behind the current failures to address biological invasions has been the strong sectoral silos between sectors that characterize policy regimes. This division has resulted in disjointed decision-making (**section 6.3.1.1**). The development of a coordinated biosecurity approach that blurs the traditional boundaries between sectors would help address environmental, health and agricultural challenges (Hulme, 2021). This cross-sector coordinated approach could provide a better way forward but is not without challenges. Collaborative, multisectoral and transdisciplinary approaches such as One Health (**Glossary**), Eco Health and One Biosecurity provide frameworks to achieve coordination between multiple sectors as well as across economies and cultures. Such coordination would facilitate the prevention and mitigation of the growing threats posed by invasive alien species. Promoting relationships between stakeholders and institutions is one option for achieving one of the key levers for transformative progress **(Figure 6.21)**. Improving policy coherence would help overcome current significant gaps in coverage of regulations targeting invasive alien species. It can also facilitate sharing efforts and commitment and understanding the specific role of all actors (**section 6.7.2.5**).

6.7.2.5 Engaging broadly across governmental sectors, industry, the scientific community, Indigenous Peoples and local communities and the wider public

General tools and approaches and frameworks exist for stakeholder engagement (**Chapter 5**, **section 5.2.1**). However, the purpose of engaging with different groups – the industrial sectors, the general public, Indigenous Peoples and local communities and the scientific community – differ **(Table 6.12)**. The design of effective engagement strategies will have context dependent elements and will take these different purposes into account (**Chapter 5**, **section 5.2.1**). The funding of engagement activities can be built into management plans and budgets to support multiple purposes in the management of biological invasions. To be effective, engagement activities can also be included within monitoring and evaluation of invasive alien species management actions, so that progress can be tracked and strategies refined over time. In this way, the effectiveness of engagement activity can also be refined and improved over time. All these elements are important for public engagement and inclusion activities to effectively contribute to implementing an integrated governance approach for biological invasions **(Figure 6.21)**.

An understanding of stakeholder and Indigenous Peoples and local communities influence and interests and how stakeholders are likely to be involved in different stages of the biological invasion process is crucial to any attempt to engage, represent, empower and co-design biological invasion management plans, and directly engage stakeholders, including citizens, as equal partners. It is clear that communicative and consultative approaches can deliver significant benefits (Shackleton, Adriaens, *et al.,* 2019). In contexts where there are significant conflicts of values or mistrust between actors, or where significant buy-in will be required to implement governance measures, it is likely to be worthwhile investing in co-productive approaches. Policy design for dealing with the anticipated impacts of invasive alien species that is sensitive to the needs and perspectives of vulnerable groups, stakeholders and Indigenous Peoples and local communities can also help achieve social justice (Blythe *et al.,* 2018; Patterson *et al.,* 2018; Temper *et al.,* 2018).

6.7.2.6 Supporting, funding and mobilizing resources for innovation, research and environmentally sound technology

(1) Improved risk assessment

There is wide variability in the capacity to respond to biological invasions amongst countries (Early *et al.,* 2016). Such variability showcases considerable imbalance in the knowledge base and implementation of best practices. Addressing such an imbalance means tracking biological invasions beyond country boundaries (Latombe *et al.,* 2017), building on the idea of connecting knowledge systems *(point 3)*. However, knowledge sharing could be paired with capacity-building and transboundary and cross-sector risk assessment tools **(Figure 6.21)**. For example, intervention strategies could focus on slowing the rates of new introductions taking place at a regional scale (e.g., African Union, European Union, MERCOSUR, USMCA/CUSMA). Thus, it is necessary to define the regions with capability deficits to determine where and how multilateral and bilateral partnerships could be forged to support those countries with limited biosecurity capabilities (Hulme, 2021). Then, within connected regions, sharing insights on invasive alien species of relevance, border control principles and methods is critical for an effective regional approach to prevention (Hulme, 2011, 2020). More generally, a coordinated and nuanced approach to risk-assessments and intervention could be extended to a global context, taking lessons from the current efforts to prevent and contain the spread of the Severe Acute Respiratory Syndrome Coronavirus 2 (SARS-CoV-2).

Table 6 12 **Engagement of stakeholders on biological invasions has a number of purposes and associated approaches and tools (options) that support the process.**

	Purpose of engagement to achieve:	Example options
1	Inclusive decision-making for biological invasions	Decision support tools (**Chapter 5**, **sections 5.2** and **5.4**), Deliberative multi-criteria analysis (**Chapter 5**, **section 5.2.2.1.j**), communication feedback systems (**section 6.3.1.4(5)**)
2	Public education and awareness raising about biological invasions	Training and risk-communication platforms (**Chapter 5**, **section 5.2.2.1** and **Table 5.6**)
3	Social learning and knowledge sharing about biological invasions; attempting to accommodate conflicting values	Co-design, co-creation and co-implementation of research and management actions (**Chapter 5**, **section 5.2.1** and **Figure 5.19**, **section 6.4.2**)
4	Coordination and collaboration for governance and management of biological invasions	Build shared trust, community-based management using adaptive management approaches (**sections 6.4.2**, **6.4.4**) and implementing appropriate, context-relevant network design for the governance structure (**sections 6.4.4.2**, **6.5.6**)
5	Surveillance and monitoring for early detection, data generation and evaluating the effectiveness of interventions for invasive alien species	Citizen science and citizen surveillance activities, including apps and data input portals (**Chapter 1**, **Figure 1.13** and **Box 1.15**; **Chapter 5**, **section 5.4.3.2** and **Table 5.6**, **section 6.6.2.1**; Box 6.20)

Developing new risk assessment tools (**Chapter 5**, **section 5.2.2.1.e**) also means employing coordinated regulatory instruments that support coherent governance for biological invasions **(Figure 6.21)** and address the fractured and disjointed approach to invasive alien species management resulting from policies that solely address issues within sectorial silos (Shine, 2007; Outhwaite, 2013; Hulme, 2021). As described in **section 6.7.2.7**, this integration could be achieved by focusing on the links between risk assessment tools of legal and regulatory instruments currently within human, animal, plant and environmental sectors.

(2) Forecasts, scenarios and models

The development of forecasting tools based on drivers that facilitate biological invasions (Essl, Lenzner, *et al.,* 2020), or mechanistic models (Sarà *et al.,* 2013; Chapman *et al.,* 2017) is vital. These forecasting tools should also focus on predicting the impacts of invasive alien species by considering the synergies in interacting drivers including invasive alien species, invasive alien species interactions and impacts (Gaertner *et al.,* 2014). These tools would then need a description of the possible scenarios of change based on shifts of the drivers in facilitating biological invasions (**Chapter 3**, **section 3.1.1**). The scenarios would describe the alternative trajectories for biological invasions within the context of complex and uncertain future socioecological developments (Alien scenarios, 2021; Roura-Pascual *et al.,* 2021).

Incorporating a wide range of modelling and scenario techniques could enable assessment of multiple pathways across spatial scales and through integration of different domains (i.e., a nexus approach; J. Liu, Hull, *et al.,* 2018). Currently joint scenario and modelling studies have a strong focus on correlative modelling approaches using single driver assessments and exploratory scenarios[19] (**Chapter 1**, **section 1.6.7.3**). Other gaps, such as the vast absence of policy-screening and target-seeking scenarios, quantifiable sustainability and policy targets for biological invasions or the widespread lack of process-based models have to be filled in order to understand the needs for and development of transformative change pathways that account for the adverse effects of invasive alien species on biodiversity, nature's contribution to people and good quality of life (**section 6.6.1.6**). Existing initiatives for transformative change related to biodiversity change (Leclère *et al.,* 2020), climate change (e.g., Burch *et al.,* 2014; Schot & Steinmueller, 2018) or food security (A. Muller *et al.,* 2017) can be taken as blueprints for the steps that can be taken to use scenarios and models to support transformative change.

19. Data management report available at: https://doi.org/10.5281/zenodo.5706520

(3) Innovative science and environmentally sound technologies to support prevention and control

The continued increasing rate in invasive alien species introductions (**Chapter 2**) highlights the need for innovative science and technologies to support the detection of and rapid response to invasive alien species (Chown *et al.,* 2015; NISC, 2016). There is a need for development of new approaches but also improvements in the effectiveness and cost-efficiencies of existing methods. In the context of technological innovations **(Figure 6.21)**, these might come from efforts or programmes focused on detection (using visual, chemical, acoustic, or genetic attributes) and/or identification for military intelligence, or human health purposes, which have not traditionally focused on invasive alien species (Martinez *et al.,* 2020; Conservation X Labs, 2017). The deployment of artificial intelligence driven internet monitoring systems for invasive species is another powerful technological advancement for the early detection of sources of known invasive alien species prior to their potential entry (Suiter & Sferrazza (2007) for an example of the application of such technology). A fundamental dimension of this development is to ensure the applicability of current technologies in diverse contexts (Martinez *et al.,* 2020; Kamenova *et al.,* 2017; **section 6.3.1.4**).

6.7.2.7 Support information systems, infrastructures and data sharing to connect knowledge systems using digital processes and international partnerships

Current understanding of the biological invasion process is adequate for taking preventive effective action. However, there remain key data, information and knowledge gaps in invasion biology and social science for bringing about widespread progress across invasion stages that could be beneficial in maximizing the efficacy of actions **(Table 6.10)**. Information systems and sharing are essential for the integrated governance for biological invasions (**Figures 6.21** and **6.23**) as these would (i) provide direction for filling key data and information gaps, (ii) enable open and equitable access to information across well and poorly resourced regions and stakeholders, (iii) facilitate the research and capacity-building needed to respond to ongoing and changing demands for information on the multiple dimensions of the problem of invasive alien species and (iv) mobilize the knowledge needed to support effective implementation of prevention and control measures (Caniglia *et al.,* 2021). Importantly, to avoid wasted investment in information platforms that collapse as resourcing changes across funding cycles, a mechanism for long-term, sustained support for an information sharing system is desirable (**sections 6.2.3.1(3)** and **6.6**).

6.7.3 Promoting a conducive environment and enabling conditions for integrated governance and transformative change

Creating a conducive environment to achieve the change that is needed is an important part of effective prevention and control of invasive alien species **(Figure 6.24)**. Good governance systems are characterized by being effective, robust, responsive and equitable (Bennett & Satterfield, 2018), and employing the strategic actions and priority interventions that bring about these characteristics **(Figure 6.24)**. These four qualities can be achieved by drawing on a broad suite of policy instruments, methods and support tools, using formal and informal decision-making structures and facilitated by processes such as negotiation, conflict resolution and knowledge sharing, particularly at local scales.

Designing, building and strengthening governance systems for biological invasions with effective, robust, responsive and equitable purposes in mind is, therefore, likely to be one of the most important determinants of progress to achieving goals for invasive alien species management. Indeed, several of the factors identified in this section, based on invasive alien species evidence, reinforce the enabling conditions identified as necessary to the 2030 mission for the Kunming Montreal Global Biodiversity Framework (CBD, 2021a), namely:

- Building participation of all stakeholders, including Indigenous Peoples and local communities;
- Inclusion of multiple sectors in decision-making;
- The need for synergies across relevant multilateral agreements and policy coherence and effectiveness monitoring;

INVASIVE ALIEN SPECIES GOVERNANCE

EFFECTIVE

Limits the spread and reduces the negative impacts of invasive alien species on nature and society by bringing about the prevention and control of invasive alien species

Levers
- Strategy and action planning for invasive alien species
- Coordination to clarify the roles, functions and mandates of actors and institutions responsible for and affected by invasive alien species
- Capacity, skills and resources, for example including strong leadership and mechanisms to resolve conflict
- Informed by evidence and a diversity of knowledge types and systems, including Indigenous and local knowledge
- Accountable, including procedures for transparent decision-making and reporting
- Efficient, for example, by prioritizing where to invest in action

RESPONSIVE

Enables adaptation to the diverse invasion stages and contexts in which invasive alien species are a concern and to the background of a changing climate

Levers
- Learning takes place across the network - from institutions to local communities - to produce, document and share information
- Anticipatory including scenario planning and supported by risk analysis and monitoring
- Adaptive so that processes to revisit, assess and evolve are institutionalized
- Innovative, enabling experimentation and higher risk tolerance, for example with appropriately regulated new technologies
- Flexible and responsive so that policies are in touch with local contexts and their diversity

EQUITABLE

Uses processes that include all relevant stakeholders and that produce fair outcomes

Levers
- Policy and processes consider and respect the diversity of perspectives, values and cultures
- Participation by having structures and spaces for engagement and collective decision making
- Mechanisms are in place to ensure fair distribution of costs and benefits, rights and responsibilities
- Just laws and policies that protect rights and provide access to justice

ROBUST

Ensures that institutions responsible for invasive alien species policy and its implementation are strong, resilient to shocks and able to maintain performance

Levers
- Legitimacy with a common vision and public support
- Connected, with strong vertical and horizontal links and relationships, supported by bridging organisations
- Nested with responsibility conferred to the lowest level possible and self-organisation encouraged
- Multiple decision-making and action centres, with some redundancy and overlap enabling resilience

Figure 6 24 **The conditions that support integrated governance for biological invasions reflecting the desirable, inter-dependent qualities of governance systems (i.e., the properties of governance systems in Figure 6.21).**

Adapted from Bennett & Satterfield (2018), https://doi.org/10.1111/conl.12600, under license CC BY 4.0.

- The establishment of cooperation mechanisms that enable collective action;
- Active involvement of sub-national and local decision-making nodes and clear assignment of roles and responsibilities;
- Preventing indirect and negative telecoupling effects (also called spill over processes), such as invasive alien species themselves;
- Recognition of the challenges at the highest levels of government and political will to act.

6.7.4 Conclusion

This chapter, supported by evidence from policy studies and other fields, and reflecting knowledge gained from previous chapters, has identified numerous options which could substantially improve invasive alien species prevention and control across multiple scales, levels of governance and sectors. It is important not to underestimate the immense threat to nature, nature's contributions to people and good quality of life posed by invasive alien species, which, at their worst are a form of persistent or irreversible pollution that can be considered as a "kind of calculable oppression" of future generations (Dasgupta, 2021; Sen, 1982). Optimal governance and policy-making conditions can be formed by policymakers, experts and citizens who are cognizant of the diverse existing approaches. Implementing integrated governance for biological invasions will only be achieved through deep cross-disciplinary discussions and planning and sustained, vigilant effort. A new focus on integrated governance stands to benefit not only invasive alien species management, but provides exciting paths towards new mechanisms and opportunities for communities to sustain good quality of life while addressing the intertwined threats to biodiversity that also threaten human civilization.

The overarching message of this chapter and of the IPBES invasive alien species assessment is clear: though there has been success in understanding and managing biological invasions, a robust, sustained and socially inclusive global commitment is necessary to avoid the most harmful impacts of invasive alien species on nature and people. The goals on reducing invasive alien species adopted by the parties to the CBD as part of the Kunming-Montreal Global Biodiversity Framework in late 2022 are attainable, but there is no time to waste in their earnest pursuit.

REFERENCES

Abrahams, B., Sitas, N., & Esler, K. J. (2019). Exploring the dynamics of research collaborations by mapping social networks in invasion science. *Journal of Environmental Management*, *229*, 27–37. https://doi.org/10.1016/j.jenvman.2018.06.051

Aceves-Bueno, E., Adeleye, A. S., Bradley, D., Tyler Brandt, W., Callery, P., Feraud, M., Garner, K. L., Gentry, R., Huang, Y., McCullough, I., Pearlman, I., Sutherland, S. A., Wilkinson, W., Yang, Y., Zink, T., Anderson, S. E., & Tague, C. (2015). Citizen Science as an Approach for Overcoming Insufficient Monitoring and Inadequate Stakeholder Buy-in in Adaptive Management: Criteria and Evidence. *Ecosystems*, *18*(3), 493–506. https://doi.org/10.1007/s10021-015-9842-4

Adams, L. B. (2014). *Victorian Rabbit Management Collaboration Initiative*. Invasive Animals Cooperative Research Centre, University of Canberra. https://pestsmart.org.au/?s=Victorian+rabbit+management+collaboration+initiative

Adams, L. B., Alter, T. R., Parkes, M. W., Reid, M., & Woolnough, A. P. (2019). Political economics, collective action and wicked socio-ecological problems: A practice story from the field. *Gateways: International Journal of Community Research and Engagement*, *12*(1). https://doi.org/10.5130/ijcre.v12i1.6496

Adams, V. M., Douglas, M. M., Jackson, S. E., Scheepers, K., Kool, J. T., & Setterfield, S. A. (2018). Conserving biodiversity and Indigenous bush tucker: Practical application of the strategic foresight framework to invasive alien species management planning. *Conservation Letters*, *11*(4), e12441. https://doi.org/10.1111/conl.12441

Adriaens, T., Sutton-Croft, M., Owen, K., Brosens, D., Valkenburg, J. van, Kilbey, D., Groom, Q., Ehmig, C., Thürkow, F., Hende, P. V., & Schneider, K. (2015). Trying to engage the crowd in recording invasive alien species in Europe: Experiences from two smartphone applications in northwest Europe. *Management of Biological Invasions*, *6*(2), 215–225. https://doi.org/10.3391/mbi.2015.6.2.12

Akerlof, G. A. (1970). The market for "lemons": Quality uncertainty and the market mechanism. *The Quarterly Journal of Economics*, *84*(3), 488–500. https://doi.org/10.2307/1879431

Alexander, J. M., Kueffer, C., Daehler, C. C., Edwards, P. J., Pauchard, A., Seipel, T., & MIREN Consortium. (2011). Assembly of nonnative floras along elevational gradients explained by directional ecological filtering. *Proceedings of the National Academy of Sciences*, *108*(2), 656–661. https://doi.org/10.1073/pnas.1013136108

Alexander, S. M., Andrachuk, M., & Armitage, D. (2016). Navigating governance networks for community-based conservation. *Frontiers in Ecology and the Environment*, *14*(3), 155–164. https://doi.org/10.1002/fee.1251

Alexander, S. M., Armitage, D., Carrington, P. J., & Bodin, Ö. (2017). Examining horizontal and vertical social ties to achieve social-ecological fit in an emerging marine reserve network. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*(6), 1209–1223. https://doi.org/10.1002/aqc.2775

Alexander, S. M., Armitage, D., & Charles, A. (2015). Social networks and transitions to co-management in Jamaican marine reserves and small-scale fisheries. *Global Environmental Change*, *35*, 213–225. https://doi.org/10.1016/j.gloenvcha.2015.09.001

Alexander, S. M., Epstein, G., Bodin, Ö., Armitage, D., & Campbell, D. (2018). Participation in planning and social networks increase social monitoring in community-based conservation. *Conservation Letters*, *11*(5), e12562. https://doi.org/10.1111/conl.12562

Alien scenarios. (2021). *Developing and applying scenarios of biological invasions for the 21st century*. https://alien-scenarios.org/

Allen, A. (2017). *Victorian rabbit action network: Impact analysis*. Victorian Rabbit Action Network. https://vran.com.au/wp-content/uploads/2020/10/VRAN-Impact-Analysis.pdf

Amano, T., Lamming, J. D. L., & Sutherland, W. J. (2016). Spatial Gaps in Global Biodiversity Information and the Role of Citizen Science. *BioScience*, *66*(5), 393–400. https://doi.org/10.1093/biosci/biw022

Amorim, F., Carvalho, S. B., Honrado, J., & Rebelo, H. (2014). Designing Optimized Multi-Species Monitoring Networks to Detect Range Shifts Driven by Climate Change: A Case Study with Bats in the North of Portugal. *PLoS ONE*, *9*(1), e87291. https://doi.org/10.1371/journal.pone.0087291

Anderies, J. M., Folke, C., Walker, B., & Ostrom, E. (2013). Aligning Key Concepts for Global Change Policy: Robustness, Resilience, and Sustainability. *Ecology and Society*, *18*(2), 8. https://doi.org/10.5751/ES-05178-180208

Anderson, C., Low-Choy, S., Whittle, P., Taylor, S., Gambley, C., Smith, L., Gillespie, P., Löcker, H., Davis, R., & Dominiak, B. (2017). Australian plant biosecurity surveillance systems. *Crop Protection*, *100*, 8–20. https://doi.org/10.1016/j.cropro.2017.05.023

Andonova, L. B., & Mitchell, R. B. (2010). The Rescaling of Global Environmental Politics. *Annual Review of Environment and Resources*, *35*(1), 255–282. https://doi.org/10.1146/annurev-environ-100809-125346

Angulo, E., & Gilna, B. (2008). When biotech crosses borders. *Nature Biotechnology*, *26*(3), 277–282. https://doi.org/10.1038/nbt0308-277

Arlidge, W. N., Bull, J. W., Addison, P. F. E., Burgass, M. J., Gianuca, D., Gorham, T. M., Jacob, C., Shumway, N., Sinclair, S. P., Watson, J. E., Wilcox, C., & Milner-Gulland, E. J. (2018). A global mitigation hierarchy for nature conservation. *BioScience*, *68*(5), 336–347. https://doi.org/10.1093/biosci/biy029

Aronson, R. B., Frederich, M., Price, R., & Thatje, S. (2015). Prospects for the return of shell-crushing crabs to Antarctica. *Journal of Biogeography*, *42*(1), 1–7. https://doi.org/10.1111/jbi.12414

Aslan, C. E., Hufford, M. B., Epanchin-Niell, R. S., Port, J. D., Sexton, J. P., & Waring, T. M. (2009). Practical Challenges in Private Stewardship of Rangeland Ecosystems: Yellow Starthistle Control in Sierra Nevadan Foothills. *Rangeland Ecology & Management*, *62*(1), 28–37. https://doi.org/10.2111/07-123

ATCM. (2006). *Practical Guidelines for Ballast Water Exchange in the Antarctic Treaty Area. Annex to Resolution 3 of 2006 agreed by the Antarctic Treaty Consultative Parties*. Antarctic Treaty Consultative Meeting. http://www.ats.aq

ATCM. (2019). *Non-Native Species Manual*. Secretariat of the Antarctic Treaty. https://documents.ats.aq/ATCM42/WW/ATCM42_WW008_e.pdf

Auerbach, N. A., Tulloch, A. I. T., & Possingham, H. P. (2014). Informed actions: Where to cost effectively manage multiple threats to species to maximize return on investment. *Ecological Applications*, *24*(6), 1357–1373. https://doi.org/10.1890/13-0711.1

August, T., Harvey, M., Lightfoot, P., Kilbey, D., Papadopoulos, T., & Jepson, P. (2015). Emerging technologies for biological recording. *Biological Journal of the Linnean Society*, *115*(3), 731–749. https://doi.org/10.1111/bij.12534

Bach, T. M., Kull, C. A., & Rangan, H. (2019). From killing lists to healthy country: Aboriginal approaches to weed control in the Kimberley, Western Australia. *Journal of Environmental Management*, *229*, 182–192. https://doi.org/10.1016/j.jenvman.2018.06.050

Bacher, S., Blackburn, T. M., Essl, F., Genovesi, P., Heikkilä, J., Jeschke, J. M., Jones, G., Keller, R., Kenis, M., Kueffer, C., Martinou, A. F., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Richardson, D. M., Roy, H. E., Saul, W.-C., Scalera, R., … Kumschick, S. (2018). Socio-economic impact classification of alien taxa (SEICAT). *Methods in Ecology and Evolution*, *9*(1), 159–168. https://doi.org/10.1111/2041-210X.12844

Bagavathiannan, M. V., Graham, S., Ma, Z., Barney, J. N., Coutts, S. R., Caicedo, A. L., De Clerck-Floate, R., West, N. M., Blank, L., Metcalf, A. L., Lacoste, M., Moreno, C. R., Evans, J. A., Burke, I., & Beckie, H. (2019). Considering weed management as a social dilemma bridges individual and collective interests. *Nature Plants*, *5*(4), 343–351. https://doi.org/10.1038/s41477-019-0395-y

Bail, C., Falkner, R., & Marquard, H. (2014). *The Cartagena Protocol on Biosafety: Reconciling trade in biotechnology with environment and development*.

Baquero, R. A., Ayllón, D., & Nicola, G. G. (2021). Are the EU biosecurity legislative frameworks sufficiently effective to prevent biological invasions in the Natura 2000 network?–A case study in Mediterranean Europe. *Environmental Science & Policy*, *120*, 21–28. https://doi.org/10.1016/j.envsci.2021.02.007

Baral, H.-O., Queloz, V., & Hosoya, T. (2014). *Hymenoscyphus fraxineus*, the correct scientific name for the fungus causing ash dieback in Europe. *IMA Fungus*, *5*(1), 79–80. https://doi.org/10.5598/imafungus.2014.05.01.09

Barbour, A., Montgomery, M., Adamson, A., Díaz-Ferguson, E., & Silliman, B. (2010). Mangrove use by the invasive lionfish *Pterois volitans*. *Marine Ecology Progress Series*, *401*, 291–294. https://doi.org/10.3354/meps08373

Barbour, R. C., Wise, S. L., McKinnon, G. E., Vaillancourt, R. E., Williamson, G. J., & Potts, B. M. (2010). The potential for gene flow from exotic eucalypt plantations into Australia's rare native eucalypts. *Forest Ecology and Management*, *260*(12), 2079–2087. https://doi.org/10.1016/j.foreco.2010.08.049

Bareghe h, A., Rowley, J., & Sambrook, S. (2009). Towards a multidisciplinary definition of innovation. *Management Decision*, *47*(8), 1323–1339. https://doi.org/10.1108/00251740910984578

Barnard, P., & Waage, J. K. (2004). *Tackling species invasions around the world: Regional responses to the invasive alien species threat*. Global Invasive Species Programme, Cape Town, South Africa. https://portals.iucn.org/library/node/28023

Barney, J. N., Schenk, T., Haak, D. C., Salom, S., Brown, B., & Hotchkiss, E. R. (2019). Building Partnerships and Bridging Science and Policy to Address the Biological Invasions Crisis. *Invasive Plant Science and Management*, *12*(1), 74–78. https://doi.org/10.1017/inp.2018.33

Barney, J. N., Tekiela, D. R., Barrios-Garcia, M. N., Dimarco, R. D., Hufbauer, R. A., Leipzig-Scott, P., Nuñez, M. A., Pauchard, A., Pyšek, P., Vítková, M., & Maxwell, B. D. (2015). Global Invader Impact Network (GIIN): Toward standardized evaluation of the ecological impacts of invasive plants. *Ecology and Evolution*, *5*(14), 2878–2889. https://doi.org/10.1002/ece3.1551

Barnhill-Dilling, S. K., Serr, M., Blondel, D. V., & Godwin, J. (2019). Sustainability as a framework for considering gene drive mice for invasive rodent eradication. *Sustainability*, *11*(5), 1334. https://doi.org/10.3390/su11051334

Baskin, Y. (2002). The greening of horticulture: New codes of conduct aim to curb plant invasions. *Bioscience*, *52*(6), 464–471. https://doi.org/10.1641/0006-3568(2002)052[0464:Tgohnc]2.0.Co;2

BBOP. (2012). *Business and Biodiversity Offsets Programme: Planning policies and projects to achieve a net gain of biodiversity* (Standard on Biodiversity Offsets). Business and Biodiversity Offsets Programme, Forest Trends. http://bbop.forest-trends.org/guidelines/Standard.pdf

Beaury, E. M., Fusco, E. J., Jackson, M. R., Laginhas, B. B., Morelli, T. L., Allen, J. M., Pasquarella, V. J., & Bradley, B. A. (2020). Incorporating climate change into invasive species management: Insights from managers. *Biological Invasions*, *22*(2), 233–252. https://doi.org/10.1007/s10530-019-02087-6

Beausoleil, N. J., Fisher, P., Littin, K. E., Warburton, B., Mellor, D. J., Dalefield, R. R., & Cowan, P. (2016). A systematic approach to evaluating and ranking the relative animal welfare impacts of wildlife control methods: Poisons used for lethal control of brushtail possums (*Trichosurus vulpecula*) in New Zealand. *Wildlife Research*, *43*(7), 553–565. https://doi.org/10.1071/WR16041

Beça, P., & Santos, R. (2010). Measuring sustainable welfare: A new approach to the ISEW. *Ecological Economics*, *69*(4), 810–819. https://doi.org/10.1016/j.ecolecon.2009.11.031

Beever, E. A., Simberloff, D., Crowley, S. L., Al-Chokhachy, R., Jackson, H. A., & Petersen, S. L. (2019). Social–ecological mismatches create conservation challenges in introduced species management. *Frontiers in Ecology and the Environment*, *17*(2), 117–125. https://doi.org/10.1002/fee.2000

Beierle, T. C. (2002). The quality of stakeholder-based decisions. *Risk Analysis: An Official Publication of the Society for Risk Analysis*, *22*(4), 739–749. https://doi.org/10.1111/0272-4332.00065

Bell, K., & Reed, M. (2022). The tree of participation: A new model for inclusive decision-making. *Community Development Journal*, *57*(4), 595–614. https://doi.org/10.1093/cdj/bsab018

Bellard, C., & Jeschke, J. M. (2016). A spatial mismatch between invader impacts and research publications. *Conservation Biology*, *30*(1), 230–232. https://doi.org/10.1111/cobi.12611

Bennett, N. J., & Satterfield, T. (2018). Environmental governance: A practical framework to guide design, evaluation, and analysis. *Conservation Letters*, *11*(6), e12600. https://doi.org/10.1111/conl.12600

Berardo, R., & Lubell, M. (2019). The ecology of games as a theory of polycentricity: Recent advances and future challenges. *Policy Studies Journal*, *47*(1), 6–26. https://doi.org/10.1111/psj.12313

Berardo, R., & Scholz, J. T. (2010). Self-Organizing Policy Networks: Risk, Partner Selection, and Cooperation in Estuaries. *American Journal of Political Science*, *54*(3), 632–649. https://doi.org/10.1111/j.1540-5907.2010.00451.x

Bertolino, S., & Genovesi, P. (2003). Spread and attempted eradication of the grey squirrel (*Sciurus carolinensis*) in Italy, and consequences for the red squirrel (*Sciurus vulgaris*) in Eurasia. *Biological Conservation*, *109*(3), 351–358. https://doi.org/10.1016/S0006-3207(02)00161-1

Best, B. D., Halpin, P. N., Fujioka, E., Read, A. J., Qian, S. S., Hazen, L. J., & Schick, R. S. (2007). Geospatial web services within a scientific workflow: Predicting marine mammal habitats in a dynamic environment. *Ecological Informatics*, *2*(3), 210–223. https://doi.org/10.1016/j.ecoinf.2007.07.007

Bhattacharyya, J., & Larson, B. M. H. (2014). The Need for Indigenous Voices in Discourse about Introduced Species: Insights from a Controversy over Wild Horses. *Environmental Values*, *23*(6), 663–684. https://doi.org/10.3197/096327114X13947900181031

Binder, C. R., Hinkel, J., Bots, P. W. G., & Pahl-Wostl, C. (2013). Comparison of Frameworks for Analyzing Social-ecological Systems. *Ecology and Society*, *18*(4), art26. https://doi.org/10.5751/ES-05551-180426

Björk, B.-C. (2017). Gold, green, and black open access. *Learned Publishing*, *30*(2), 173–175. https://doi.org/10.1002/leap.1096

Blackburn, T. M., Bellard, C., & Ricciardi, A. (2019). Alien *versus* native species as drivers of recent extinctions. *Frontiers in Ecology and the Environment*, *17*(4), 203–207. https://doi.org/10.1002/fee.2020

Blackman, A., & Rivera, J. (2010). *Environmental Certification and the Global Environment Facility: A STAP advisory document*. Scientific and Technical Advisory Panel (STAP) of the Global Environment Facility (GEF). https://www.thegef.org/sites/default/files/council-meeting-documents/C.39.Inf_.15_STAP_-_Environmental_Certification.pdf

Blaser, W. J. (2013). *Impact of woody encroachment on soil-plant-herbivore interactions in the Kafue Flats floodplain ecosystem* [Doctoral Thesis, ETH Zurich]. https://doi.org/10.3929/ethz-a-009933926

Blythe, J., Silver, J., Evans, L., Armitage, D., Bennett, N. J., Moore, M.-L., Morrison, T. H., & Brown, K. (2018). The dark side of transformation: Latent risks in contemporary sustainability discourse. *Antipode*, *50*(5), 1206–1223. https://doi.org/10.1111/anti.12405

Boakes, E. H., McGowan, P. J. K., Fuller, R. A., Chang-qing, D., Clark, N. E., O'Connor, K., & Mace, G. M. (2010). Distorted Views of Biodiversity: Spatial and Temporal Bias in Species Occurrence Data. *PLoS Biology*, *8*(6), e1000385. https://doi.org/10.1371/journal.pbio.1000385

Bodin, Ö. (2017). Collaborative environmental governance: Achieving collective action in social-ecological systems. *Science*, *357*(6352), eaan1114. https://doi.org/10.1126/science.aan1114

Bodin, Ö., Alexander, S. M., Baggio, J., Barnes, M. L., Berardo, R., Cumming, G. S., Dee, L. E., Fischer, A. P., Fischer, M., Mancilla Garcia, M., Guerrero, A. M., Hileman, J., Ingold, K., Matous, P., Morrison, T. H., Nohrstedt, D., Pittman, J., Robins, G., & Sayles, J. S. (2019). Improving network approaches to the study of complex social–ecological interdependencies. *Nature Sustainability*, *2*(7), 551–559. https://doi.org/10.1038/s41893-019-0308-0

Bodin, Ö., Baird, J., Schultz, L., Plummer, R., & Armitage, D. (2020). The impacts of trust, cost and risk on collaboration in environmental governance. *People and Nature*, *2*(3), 734–749. https://doi.org/10.1002/pan3.10097

Bodin, Ö., Sandström, A., & Crona, B. (2017). Collaborative Networks for Effective Ecosystem-Based Management: A Set of Working Hypotheses. *Policy Studies Journal*, *45*(2), 289–314. https://doi.org/10.1111/psj.12146

Bohannon, J. (2016). Who's downloading pirated papers? Everyone. *Science*, *352*(6285), 508–512. https://doi.org/10.1126/science.352.6285.508

Bohensky, E. L., & Maru, Y. (2011). Indigenous Knowledge, Science, and Resilience: What Have We Learned from a Decade of International Literature on "Integration"? *Ecology and Society*, *16*(4), 6. https://doi.org/10.5751/ES-04342-160406

Bolhassan, R., Cranefield, J., & Dorner, D. (2014). Indigenous Knowledge Sharing in Sarawak: A System-Level View and Its Implications for the Cultural Heritage Sector. *2014 47th Hawaii International Conference on System Sciences*, 3378–3388. https://doi.org/10.1109/HICSS.2014.419

Boon, P. J., Clarke, S. A., & Copp, G. H. (2020). Alien species and the EU Water Framework Directive: A comparative assessment of European approaches. *Biological Invasions*, *22*(4), 1497–1512. https://doi.org/10.1007/s10530-020-02201-z

Boström, M., Grönholm, S., & Hassler, B. (2016). The Ecosystem Approach to Management in Baltic Sea Governance: Towards Increased Reflexivity? In M. Gilek, M. Karlsson, S. Linke, & K. Smolarz (Eds.), *Environmental Governance of the Baltic Sea* (Vol. 10, pp. 149–172). Springer. https://doi.org/10.1007/978-3-319-27006-7_7

Bradley, B. A., & Wilcove, D. S. (2009). When Invasive Plants Disappear: Transformative Restoration Possibilities in the Western United States Resulting from Climate Change. *Restoration Ecology*, *17*(5), 715–721. https://doi.org/10.1111/j.1526-100X.2009.00586.x

Brauman, K. A., Garibaldi, L. A., Polasky, S., Zayas, C. N., Aumeeruddy Thomas, Y., Brancalion, P., Declerck, F., Mastrangelo, M., Nkongolo, N., Palang, H., Shannon, L., Verma, M., & Shrestha, U. B. (2020). *IPBES Global Assessment Report on Biodiversity and Ecosystem Services Chapter 2.3. Status and Trends -Nature's Contributions to People (NCP)*. Zenodo. https://doi.org/10.5281/ZENODO.3832035

Braun, H. A., Kowalski, K. P., & Hollins, K. (2016). Applying the collective impact approach to address non-native species: A case study of the Great Lakes Phragmites Collaborative. *Biological Invasions*, *18*(9), 2729–2738. https://doi.org/10.1007/s10530-016-1142-1

Bravo-Vargas, V., García, R. A., Pizarro, J. C., & Pauchard, A. (2019). Do people care about pine invasions? Visitor perceptions and willingness to pay for pine control in a protected area. *Journal of Environmental Management*, *229*(December 2017), 57–66. https://doi.org/10.1016/j.jenvman.2018.07.018

Brazeiro, A., Bresciano, D., Brugnoli, E., & Iturburu, M. (2021). *Especies exóticas invasorasde Uruguay: Distribución, impactos socioambientales y estrategias de gestión.* (Montevideo, p. 226). RETEMA-UdelaR, CEEI-Ministerio de Ambiente. https://ppduruguay.undp.org.uy/wp-content/uploads/2022/06/Especies-Exoticas-Invasoras-en-Uruguay_compressed.pdf

Brenton-Rule, E. C., Barbieri, R. F., & Lester, P. J. (2016). *Data from: Corruption, development and governance indicators predict invasive species risk from trade* [dataset]. Dryad. https://doi.org/10.5061/dryad.1qq64

Broderstad, E. G., & Eythórsson, E. (2014). Resilient communities? Collapse and recovery of a social-ecological system in Arctic Norway. *Ecology and Society*, *19*(3), 1. https://doi.org/10.5751/ES-06533-190301

Brown, P. M. J., Roy, D. B., Harrower, C., Dean, H. J., Rorke, S. L., & Roy, H. E. (2018). Spread of a model invasive alien species, the harlequin ladybird *Harmonia axyridis* in Britain and Ireland. *Scientific Data*, *5*(1), 180239. https://doi.org/10.1038/sdata.2018.239

Brugnoli, E., & Laufer, G. (Eds.). (2018). *Ecología, Manejo y Control de Especies Exóticas Invasoras en Uruguay, del Diagnóstico a la Acción*. Comité de Especies Exóticas Invasoras.

Brundu, G., Pauchard, A., Pyšek, P., Pergl, J., Bindewald, A., Brunori, A., Canavan, S., Campagnaro, T., Celesti-Grapow, L., de Sá Dechoum, M., Dufour-Dror, J.-M., Essl, F., Flory, S. L., Genovesi, P., Guarino, F., Guangzhe, L., Hulme, P. E., Jäger, H., Kettle, C. J., … Richardson, D. M. (2020). Global guidelines for the sustainable use of non-native trees to prevent tree invasions and mitigate their negative impacts. *NeoBiota*, *61*, 65–116. https://doi.org/10.3897/neobiota.61.58380

Brundu, G., & Richardson, D. M. (2016). Planted forests and invasive alien trees in Europe: A code for managing existing and future plantings to mitigate the risk of negative impacts from invasions. *NeoBiota*, *30*, 5–47. https://doi.org/10.3897/neobiota.30.7015

Bryce, R., Oliver, M. K., Davies, L., Gray, H., Urquhart, J., & Lambin, X. (2011). Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management. *Biological Conservation*, *144*(1), 575–583. https://doi.org/10.1016/j.biocon.2010.10.013

Buchanan, G. M., Butchart, S. H., Chandler, G., & Gregory, R. D. (2020). Assessment of national-level progress towards elements of the Aichi Biodiversity Targets. *Ecological Indicators*, *116*, 106497. https://doi.org/10.1016/j.ecolind.2020.106497

Buck, M., & Hamilton, C. (2011). The Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization to the Convention on Biological Diversity. *Review of European Community & International Environmental Law*, *20*(1), 47–61. https://doi.org/10.1111/j.1467-9388.2011.00703.x

Budapest Open Access Initiative. (2012). *Budapest Open Access Initiative*. https://www.budapestopenaccessinitiative.org/

Burch, S., Shaw, A., Dale, A., & Robinson, J. (2014). Triggering transformative change: A development path approach to climate change response in communities. *Climate Policy*, *14*(4), 467–487. https://doi.org/10.1080/14693062.2014.876342

Burgess, H. K., DeBey, L. B., Froehlich, H. E., Schmidt, N., Theobald, E. J., Ettinger, A. K., HilleRisLambers, J., Tewksbury, J., & Parrish, J. K. (2017). The science of citizen science: Exploring barriers to use as a primary research tool. *Biological Conservation*, *208*, 113–120. https://doi.org/10.1016/j.biocon.2016.05.014

Burgiel, S. W. (2015). From global to local: Integrating policy frameworks for the prevention and management of invasive species. In R. P. Keller, M. W. Cadotte, & G. Sandiford (Eds.), *Invasive Species in a Globalized World: Ecological, Social, And Legal Perspectives On Policy* (pp. 283–302). University of Chicago Press. https://doi.org/10.7208/chicago/9780226166216.003.0013

Burke, G., Singh, B. R., & Theodore, L. (2005). *Handbook of environmental management and technology.* (Issue Ed. 2). John Wiley & Sons, Inc.

Butchart, S. H. M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J. P. W., Almond, R. E. A., Baillie, J. E. M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K. E., Carr, G. M., Chanson, J., Chenery, A. M., Csirke, J., Davidson, N. C., Dentener, F., Foster, M., Galli, A., … Watson, R. (2010). Global Biodiversity: Indicators of Recent Declines. *Science*, *328*(5982), 1164–1168. https://doi.org/10.1126/science.1187512

Byrne, M. J., du Plessis, D., Ivey, P. J., Measey, J., Robertson, M. P., Robinson, T. B., & Weaver, K. N. (2020). Education, training and capacity-building in the field of biological invasions in South Africa. In B. W. van Wilgen, J. Measey, D. M. Richardson, J. R. Wilson, & T. A. Zengeya (Eds.), *Biological Invasions in South Africa* (Vol. 14, pp. 731–755). Springer. https://doi.org/10.1007/978-3-030-32394-3_25

Calviño-Cancela, M., & Rubido-Bará, M. (2013). Invasive potential of *Eucalyptus globulus*: Seed dispersal, seedling recruitment and survival in habitats surrounding plantations. *Forest Ecology and Management*, *305*, 129–137. https://doi.org/10.1016/j.foreco.2013.05.037

Camia, A., Giuntoli, J., Jonsson, R., Robert, N., Cazzaniga, N. E., Jasinevičius, G., Avitabile, V., Grassi, G., Barredo, J. I., & Mubareka, S. (2021). *The use of woody biomass for energy production in the EU* (EUR 30548). Publications Office of the European Union. https://doi.org/10.2760/831621

Canhos, V. P., Souza, S. D., Giovanni, R. D., & Canhos, D. A. L. (2004). Global Biodiversity Informatics: Setting the scene for a "new world" of ecological forecasting. *Biodiversity Informatics*, *1*. https://doi.org/10.17161/bi.v1i0.3

Caniglia, G., Luederitz, C., von Wirth, T., Fazey, I., Martin-López, B., Hondrila, K., König, A., von Wehrden, H., Schäpke, N. A., Laubichler, M. D., & Lang, D. J. (2021). A pluralistic and integrated approach to action-oriented knowledge for sustainability. *Nature Sustainability*, *4*(2), 93–100. https://doi.org/10.1038/s41893-020-00616-z

Carlton, J. T. (1996). Biological Invasions and Cryptogenic Species. *Ecology*, *77*(6), 1653–1655. https://doi.org/10.2307/2265767

Carson, R. (1962). *Silent Spring*. Houghton Mifflin. https://www.environmentandsociety.org/mml/silent-spring

Carter, L., Mankad, A., Zhang, A., Curnock, M. I., & Pollard, C. R. J. (2021). A multidimensional framework to inform stakeholder engagement in the science and management of invasive and pest animal species. *Biological Invasions*, *23*(2), 625–640. https://doi.org/10.1007/s10530-020-02391-6

Carthey, A. J. R., & Banks, P. B. (2012). When Does an Alien Become a Native Species? A Vulnerable Native Mammal Recognizes and Responds to Its Long-Term Alien Predator. *PLoS ONE*, *7*(2), e31804. https://doi.org/10.1371/journal.pone.0031804

Cason, T. N., & Gangadharan, L. (2002). Environmental labeling and incomplete consumer information in laboratory markets. *Journal of Environmental Economics and Management*, *43*(1), 113–134. https://doi.org/10.1006/jeem.2000.1170

Catacutan, D., Neely, C., Johnson, M., Poussard, H., & Youl, R. (2009).

Landcare:Local action – global progress. World Agroforestry Centre. http://apps.worldagroforestry.org/downloads/Publications/PDFS/B16017.pdf

CBD. (2000). *Cartagena Protocol on Biosafety to the Convention on Biological Diversity: Text and Annexes*. Secretariat of the Convention on Biological Diversity. https://wedocs.unep.org/20.500.11822/27553

CBD. (2011). *Incentive measures for the conservation and sustainable use of biological diversity. Case studies and lessons learned* (56; CBD Technical Series, Issue 56). Secretariat of the Convention on Biological Diversity. https://www.cbd.int/doc/publications/cbd-ts-56-en.pdf

CBD. (2014a). *Pathways of introduction of invasive species, their prioritization and management*. 18. https://www.cbd.int/doc/meetings/sbstta/sbstta-18/official/sbstta-18-09-add1-en.pdf

CBD. (2014b). *XII/17. Invasive alien species: Review of work and considerations for future work* (UNEP/CBD/COP/DEC/XII/17) [Decision Adopted By The Conference Of The Parties To The Convention On Biological Diversity]. https://www.cbd.int/doc/decisions/cop-12/cop-12-dec-17-en.pdf

CBD. (2020a). *Zero draft of the post-2020 Global Biodiversity Framework* (Open Ended Working Group on The Post-2020 Global Biodiversity Framework CBD/WG2020/2/3; p. 16). Convention on Biodiversity, United Nations Environment Programme. https://www.cbd.int/doc/c/efb0/1f84/a892b98d2982a829962b6371/wg2020-02-03-en.pdf

CBD. (2021a). *CBD/WG2020/3/3: First draft of the post-2020 global biodiversity framework*. Convention on Biodiversity, United Nations Environment Programme. https://www.cbd.int/doc/c/abb5/591f/2e46096d3f0330b08ce87a45/wg2020-03-03-en.pdf

CBD. (2021b). *Estimation of Resources Needed for Implementing the Post-2020 Global Biodiversity Framework* (Subsidiary Body on Implementation Third Meeting CBD/SBI/3/5/Add.2/Rev.1). Convention on Biodiversity, United Nations Environment Programme. https://www.cbd.int/doc/c/b858/143f/c58220e9a61c4b5fc2dfeed3/sbi-03-05-add2-rev1-en.pdf

CBD. (2002, April 7). Alien species that threaten ecosystems, habitats or species. *COP 6 Decision VI/23*. COP 6, The Hague, Netherlands. https://www.cbd.int/decision/cop/?id=7197

CBD. (2012). XI/28. Invasive alien species. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Eleventh meeting of Conference of the parties to the Convention on Biological Diversity, Hyderabad, India. https://www.cbd.int/doc/decisions/cop-11/cop-11-dec-28-en.pdf

CBD. (2018). 14/11. Invasive alien species. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Fourteen meeting of Conference of the parties to the Convention on Biological Diversity, Sharm El-Sheikh, Egypt. https://www.cbd.int/doc/decisions/cop-14/cop-14-dec-11-en.pdf

CBD. (2020b). *Preparations for the Post-2020 Biodiversity Framework: Fifth meeting of the Open-ended Working Group on the Post-2020 Global Biodiversity Framework*. Convention on Biological Diversity. https://www.cbd.int/conferences/post2020

CBD. (2020c). *Strategic Plan for Biodiversity 2011-2020, including Aichi Biodiversity Targets*. Convention on Biological Diversity; Secretariat of the Convention on Biological Diversity. https://www.cbd.int/sp/

CBD. (2022a). 15/4. Kunming-Montreal Global Biodiversity Framework. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Fifteenth meeting (Part II) of Conference of the parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf

CBD. (2022b). 15/5. Monitoring framework for the Kunming-Montreal Global Biodiversity Framework. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Fifteenth meeting (Part II) of Conference of the parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-05-en.pdf

CBD. (2022c). 15/27. Invasive alien species. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity*. Fifteenth meeting of Conference of the parties to the Convention on Biological Diversity, Montreal, Canada. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-27-en.pdf

Chaffin, B. C., Garmestani, A. S., Angeler, D. G., Herrmann, D. L., Stow, C. A., Nyström, M., Sendzimir, J., Hopton, M. E., Kolasa, J., & Allen, C. R. (2016). Biological invasions, ecological resilience and adaptive governance. *Journal of Environmental Management*, *183*(Pt 2), 399–407. https://doi.org/10.1016/j.jenvman.2016.04.040

Chaffin, B. C., Gosnell, H., & Cosens, B. A. (2014). A decade of adaptive governance scholarship: Synthesis and future directions. *Ecology and Society*, *19*(3), 56. https://doi.org/10.5751/ES-06824-190356

Chan, K. M. A., Boyd, D. R., Gould, R. K., Jetzkowitz, J., Liu, J., Muraca, B., Naidoo, R., Olmsted, P., Satterfield, T., Selomane, O., Singh, G. G., Sumaila, R., Ngo, H. T., Boedhihartono, A. K., Agard, J., Aguiar, A. P. D., Armenteras, D., Balint, L., Barrington-Leigh, C., … Brondízio, E. S. (2020). Levers and leverage points for pathways to sustainability. *People and Nature*, *2*(3), 693–717. https://doi.org/10.1002/pan3.10124

Chandler, M., See, L., Copas, K., Bonde, A. M. Z., López, B. C., Danielsen, F., Legind, J. K., Masinde, S., Miller-Rushing, A. J., Newman, G., Rosemartin, A., & Turak, E. (2017). Contribution of citizen science towards international biodiversity monitoring. *Biological Conservation*, *213*, 280–294. https://doi.org/10.1016/J.BIOCON.2016.09.004

Chapman, D. S., Scalone, R., Štefanić, E., & Bullock, J. M. (2017). Mechanistic species distribution modeling reveals a niche shift during invasion. *Ecology*, *98*(6), 1671–1680. https://doi.org/10.1002/ecy.1835

Chown, S. L., & Brooks, C. M. (2019). The state and future of Antarctic environments in a global context. *Annual Review of Environment and Resources*, *44*, 1–30. https://doi.org/10.1146/annurev-environ-101718-033236

Chown, S. L., Brooks, C. M., Terauds, A., Le Bohec, C., van Klaveren-Impagliazzo, C., Whittington, J. D., Butchart, S. H., Coetzee, B. W. T., Collen, B., Convey, P., Gaston, K. J., Gilbert, N., Gill, M., Höft, R., Johnston, S., Kennicutt II, M. C., Kriesell, H. J., Le Maho, Y., Lynch, H. J., … McGeoch, M. A. (2017). Antarctica and the Strategic Plan for Biodiversity. *PLoS Biology*, *15*(3), e2001656. https://doi.org/10.1371/journal.pbio.2001656

Chown, S. L., Hodgins, K. A., Griffin, P. C., Oakeshott, J. G., Byrne, M., & Hoffmann, A. A. (2015). Biological invasions, climate change and genomics. *Evolutionary Applications*, *8*(1), 23–46. https://doi.org/10.1111/eva.12234

CITES. (1975). *Convention on International Trade in Endangered Species of Wild Fauna and Flora*. Convention on International Trade in Endangered Species of Wild Fauna and Flora. https://www.cites.org/eng/disc/text.php

Clark, W. C., & Harley, A. G. (2020). Sustainability Science: Toward a Synthesis. *Annual Review of Environment and Resources*, *45*(1), 331–386. https://doi.org/10.1146/annurev-environ-012420-043621

Climate Action Tracker. (2020). *Paris Agreement turning point: Wave of net zero targets reduces warming estimate to 2.1 degree celcius in 2100 All eyes on 2030 targets*. NewClimate Institute and Climate Analytics. https://climateactiontracker.org/documents/829/CAT_2020-12-01_Briefing_GlobalUpdate_Paris5Years_Dec2020.pdf

CMS. (1979). *Convention on the Conservation of Migratory Species of Wild Animals*. Convention on the Conservation of Migratory Species of Wild Animals. https://www.cms.int/sites/ default/files/instrument/CMS-text.en_.PDF

Coglianese, C. (1997). Assessing consensus: The promise and performance of negotiated rulemaking. *Duke Law Journal*, *46*(6), 1255. https://doi.org/10.2307/1372989

Collins, J. P. (2018). Gene drives in our future: Challenges of and opportunities for using a self-sustaining technology in pest and vector management. *BMC Proceedings*, *12*(Suppl 8), 9. https://doi.org/10.1186/s12919-018-0110-4

Commission of the European Communities. (2009). *COM(2009)163 final. Green Paper—Reform of the Common Fisheries Policy*. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2009:0163:FIN:EN:PDF

Conservation X Labs. (2017). *The innovation summit: Overcoming the invasive species challenge*. National Invasive Species Council. https://on.doi.gov/2pmGXCp

Cook, D. C., Kristensen, N. P., Liu, S., Paini, D. R., Kerr, P. J., Sheppard, A. W., Lonsdale, W. M., McAllister, R. R. J., & De Barro, P. J. (2014). Plant biosecurity policy-making modelled on the human immune system: What would it look like? *Environmental Science & Policy*, *41*, 1–10. https://doi.org/10.1016/j.envsci.2014.04.007

Cook, D. C., Liu, S., Murphy, B., & Lonsdale, W. M. (2010). Adaptive Approaches to Biosecurity Governance. *Risk Analysis*, *30*(9), 1303–1314. https://doi.org/10.1111/j.1539-6924.2010.01439.x

Cook, W., van Bommel, S., & Turnhout, E. (2016). Inside environmental auditing: Effectiveness, objectivity, and transparency. *Current Opinion in Environmental Sustainability*, *18*, 33–39. https://doi.org/10.1016/j.cosust.2015.07.016

Cooke, B., & Kothari, U. (2001). *Participation: The new tyranny?* Zed Books.

Cooney, R., & Lang, A. T. F. (2007). Taking uncertainty seriously: Adaptive governance and international trade. *European Journal of International Law*, *18*(3), 523–551. https://doi.org/10.1093/ejil/chm030

Côté, I. M., Green, S. J., & Hixon, M. A. (2013). Predatory fish invaders: Insights from Indo-Pacific lionfish in the western Atlantic and Caribbean. *Biological Conservation*, *164*, 50–61. https://doi.org/10.1016/j.biocon.2013.04.014

Council of Europe. (1993). *Convention on Civil Liability for Damage Resulting from Activities Dangerous to the Environment* (150; European Treaty Series, p. 16). Council of Europe.

Council of the European Communities. (2008). *Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). OJEU L164:19–40*. https://faolex.fao.org/docs/pdf/eur80075.pdf

Council of the European Communities. (2014). *Regulation (EU) No 1143/2014 of the European Parliament and of the Council of 22 October 2014 on the prevention and management of the introduction and spread of invasive alien species. OJEU L317:35–55*. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=OJ:JOL_2014_317_R_0003

Crall, A. W., Jarnevich, C. S., Young, N. E., Panke, B. J., Renz, M., & Stohlgren, T. J. (2015). Citizen science contributes to our knowledge of invasive plant species distributions. *Biological Invasions*, *17*(8), 2415–2427. https://doi.org/10.1007/s10530-015-0885-4

Crall, A. W., Newman, G. J., Stohlgren, T. J., Holfelder, K. A., Graham, J., & Waller, D. M. (2011). Assessing citizen science data quality: An invasive species case study. *Conservation Letters*, *4*(6), 433–442. https://doi.org/10.1111/j.1755-263X.2011.00196.x

Creighton, M. J. A., & Bennett, J. R. (2019). Taxonomic biases persist from listing to management for Canadian species at risk. *Écoscience*, *26*(4), 315–321. https://doi.org/10.1080/11956860.2019.1613752

Crona, B. I., & Parker, J. N. (2012). Learning in Support of Governance: Theories, Methods, and a Framework to Assess How Bridging Organizations Contribute to Adaptive Resource Governance. *Ecology and Society*, *17*(1), 32. https://doi.org/10.5751/ES-04534-170132

Crowley, S. L., Hinchliffe, S., & McDonald, R. A. (2017a). Conflict in invasive species management. *Frontiers in Ecology and the Environment*, *15*(3), 133–141. https://doi.org/10.1002/fee.1471

Crowley, S. L., Hinchliffe, S., & McDonald, R. A. (2017b). Invasive species management will benefit from social impact assessment. *Journal of Applied Ecology*, *54*(2), 351–357. https://doi.org/10.1111/1365-2664.12817

Crowley, S. L., Hinchliffe, S., & McDonald, R. A. (2019). The parakeet protectors: Understanding opposition to introduced species management. *Journal of Environmental Management*, *229*, 120–132. https://doi.org/10.1016/j.jenvman.2017.11.036

CSIRO. (2022). *Australia's Biosecurity Future: Unlocking the next decade of resilience*. Commonwealth Scientific and Industrial Research Organisation. https://www.csiro.au/en/work-with-us/services/consultancy-strategic-advice-services/csiro-futures/agriculture-and-food/biosecurity-futures

Čuda, J., Vitkova, M., Albrechtová, M., Guo, W.-Y., Barney, J. N., & Pyšek, P. (2017). Invasive herb *Impatiens glandulifera* has minimal impact on multiple components of temperate forest ecosystem function. *Biological Invasions*, *19*, 3051–3066. https://doi.org/10.1007/s10530-017-1508-z

Culley, T. M., & Hardiman, N. A. (2007). The Beginning of a New Invasive Plant: A History of the Ornamental Callery Pear in the United States. *BioScience*, *57*(11), 956–964. https://doi.org/10.1641/B571108

Culley, T. M., & Hardiman, N. A. (2009). The role of intraspecific hybridization in the evolution of invasiveness: A case study of the ornamental pear tree *Pyrus calleryana*. *Biological Invasions*, *11*(5), 1107–1119. https://doi.org/10.1007/s10530-008-9386-z

Culley, T. M., Hardiman, N. A., & Hawks, J. (2011). The role of horticulture in plant invasions: How grafting in cultivars of Callery pear (*Pyrus calleryana*) can facilitate spread into natural areas. *Biological Invasions*, *13*(3), 739–746. https://doi.org/10.1007/s10530-010-9864-y

Cunningham, S., Teirney, L., Brunton, J., McLeod, R., Bowman, R., Richards, D., Kinsey, R., & Matthews, F. (2019). Mitigating the threat of invasive marine species to Fiordland: New Zealand's first pathway management plan. *Management of Biological Invasions*, *10*(4), 690–708. https://doi.org/10.3391/mbi.2019.10.4.07

D'Amen, M., & Azzurro, E. (2020). Lessepsian fish invasion in Mediterranean marine protected areas: A risk assessment under climate change scenarios. *ICES Journal of Marine Science*, *77*(1), 388–397. https://doi.org/10.1093/icesjms/fsz207

Dandy, N., Marzano, M., Porth, E. F., Urquhart, J., & Potter, C. (2017). Who has a stake in ash dieback? A conceptual framework for the identification and categorisation of tree health stakeholders. In R. Vasaitis & R. Enderle (Eds.), *Dieback of European Ash (Fraxinus spp.)* (pp. 15–26). Swedish University of Agricultural Sciences. https://www.cabi.org/isc/abstract/20183360941

Darling, J. A., Galil, B. S., Carvalho, G. R., Rius, M., Viard, F., & Piraino, S. (2017). Recommendations for developing and applying genetic tools to assess and manage biological invasions in marine ecosystems. *Marine Policy*, *85*, 54–64. https://doi.org/10.1016/j.marpol.2017.08.014

Dasgupta, P. (2021). *The Economics of Biodiversity:The Dasgupta Review. Abridged Version* (p. 103). https://www.gov.uk/government/publications/the-economics-of-biodiversity-the-dasgupta-review-abridged-version-translations

Daviter, F. (2017). Coping, taming or solving: Alternative approaches to the governance of wicked problems. *Policy Studies*, *38*(6), 571–588. https://doi.org/10.1080/01442872.2017.1384543

de Almeida, C. de F. C. B. R., Ramos, M. A., de Amorim, E. L. C., & de Albuquerque, U. P. (2010). A comparison of knowledge about medicinal plants for three rural communities in the semi-arid region of northeast of Brazil. *Journal of Ethnopharmacology*, *127*(3), 674–684. https://doi.org/10.1016/j.jep.2009.12.005

De Giovanni, R., Williams, A. R., Ernst, V. H., Kulawik, R., Fernandez, F. Q., & Hardisty, A. R. (2016). ENM Components: A new set of web service-based workflow components for ecological niche modelling. *Ecography*, *39*(4), 376–383. https://doi.org/10.1111/ecog.01552

De Klemm, C. (1996). *Introductions of non-native organisms into the natural environment* (Issues 18–73). Council of Europe.

De Silva, S. S. (2012). Aquaculture: A newly emergent food production sector—and perspectives of its impacts on biodiversity and conservation. *Biodiversity and Conservation*, *21*(12), 3187–3220. https://doi.org/10.1007/s10531-012-0360-9

De Silva, S. S., Nguyen, T. T. T., Abery, N. W., & Amarasinghe, U. S. (2006). An evaluation of the role and impacts of alien finfish in Asian inland aquaculture. *Aquaculture Research*, *37*(1), 1–17. https://doi.org/10.1111/j.1365-2109.2005.01369.x

De Silva, S. S., Nguyen, T. T. T., Turchini, G. M., Amarasinghe, U. S., & Abery, N. W. (2009). Alien Species in Aquaculture and Biodiversity: A Paradox in Food Production. *AMBIO: A Journal of the Human Environment*, *38*(1), 24–28. https://doi.org/10.1579/0044-7447-38.1.24

De Silva, S. S., Subasinghe, R., Bartley, D. M., & Lowther, A. (2004). *Tilapias as alien aquatics in Asia and the Pacific: A review* (FAO Fisheries Technical Paper No. 453). Food and Agriculture Organization.

De Vente, J., Reed, M. S., Stringer, L. C., Valente, S., & Newig, J. (2016). How does the context and design of participatory decision making processes affect their outcomes? Evidence from sustainable land management in global drylands. *Ecology and Society*, *21*(2), 24. https://doi.org/10.5751/ES-08053-210224

DeFries, R., & Nagendra, H. (2017). Ecosystem management as a wicked problem. *Science*, *356*(6335), 265–270. https://doi.org/10.1126/science.aal1950

Dehnen-Schmutz, K. (2011). Determining non-invasiveness in ornamental plants to build green lists. *Journal of Applied Ecology*, *48*(6), 1374–1380. https://doi.org/10.1111/j.1365-2664.2011.02061.x

Dehnen-Schmutz, K., Holdenrieder, O., Jeger, M. J., & Pautasso, M. (2010). Structural change in the international horticultural industry: Some implications for plant health. *Scientia Horticulturae*, *125*(1), 1–15. https://doi.org/10.1016/j.scienta.2010.02.017

Delaney, D. G., Sperling, C. D., Adams, C. S., & Leung, B. (2008). Marine invasive species: Validation of citizen science and implications for national monitoring networks. *Biological Invasions*, *10*(1), 117–128. https://doi.org/10.1007/s10530-007-9114-0

Derak, M., Cortina, J., Taiqui, L., & Aledo, A. (2018). A proposed framework for participatory forest restoration in semiarid areas of North Africa. *Restoration Ecology*, *26*(S1), S18–S25. https://doi.org/10.1111/rec.12486

Derraik, J. G. B., & Phillips, S. (2010). Online trade poses a threat to biosecurity in New Zealand. *Biological Invasions*, *12*(6), 1477–1480. https://doi.org/10.1007/s10530-009-9595-0

Deutz, A., Heal, G. M., Niu, R., Swanson, E., Townshend, T., Zhu, L., Delmar, A., Meghji, A., Sethi, S. A., & Tobin-de la Puente, J. (2020). *Financing Nature: Closing the Global Biodiversity Financing Gap* [Full Report]. The Paulson Institute, The Nature Conservancy, and the Cornell Atkinson Center for Sustainability. https://www.paulsoninstitute.org/wp-content/uploads/2020/09/FINANCING-NATURE_Full-Report_Final-Version_091520.pdf

Diagne, C., Leroy, B., Gozlan, R. E., Vaissière, A.-C., Assailly, C., Nuninger, L., Roiz, D., Jourdain, F., Jarić, I., & Courchamp, F. (2020). InvaCost, a public database of the economic costs of biological invasions worldwide. *Scientific Data*, *7*(1), 277. https://doi.org/10.1038/s41597-020-00586-z

Diagne, C., Leroy, B., Vaissière, A.-C., Gozlan, R. E., Roiz, D., Jarić, I., Salles, J.-M., Bradshaw, C. J. A., & Courchamp, F. (2021). High and rising economic costs of biological invasions worldwide. *Nature*, *592*(7855), 571–576. https://doi.org/10.1038/s41586-021-03405-6

Díaz, C. L. (2010). The Bern convention: 30 years of nature conservation in Europe. *Review of European Community and International Environmental Law*, *19*(2), 185–196. https://doi.org/10.1111/j.1467-9388.2010.00676.x

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J. R., Arico, S., Báldi, A., Bartuska, A., Baste, I. A., Bilgin, A., Brondizio, E., Chan, K. M., Figueroa, V. E., Duraiappah, A., Fischer, M., Hill, R., … Zlatanova, D. (2015). The IPBES Conceptual Framework—Connecting nature and people. *Current Opinion in Environmental Sustainability*, *14*, 1–16. https://doi.org/10.1016/j.cosust.2014.11.002

Díaz, S., Settele, J., Brondízio, E. S., Ngo, H. T., Agard, J., Arneth, A., Balvanera, P., Brauman, K. A., Butchart, S. H. M., Chan, K. M. A., Garibaldi, L. A., Ichii, K., Liu, J., Subramanian, S. M., Midgley, G. F., Miloslavich, P., Molnár, Z., Obura, D., Pfaff, A., … Zayas, C. N. (2019). Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science*, *366*(6471), eaax3100. https://doi.org/10.1126/science.aax3100

Díaz-Ferguson, E. E. (2012). *Introducción a la ecología molecular marina: Aplicaciones y perspectivas* (1st ed.). Universal Books.

Díaz-Ferguson, E. E., & Hunter, M. E. (2019). Life history, genetics, range expansion and new frontiers of the lionfish (*Pterois volitans*, Perciformes: Pteroidae) in Latin America. *Regional Studies in Marine Science*, *31*, 100793. https://doi.org/10.1016/j.rsma.2019.100793

Díaz-Ferguson, E. E., & Moyer, G. R. (2014). History, applications and methodological issues and perspectives for the use of environmental DNA (eDNA) in marine and freshwater environments. *Revista de Biología Tropical*, *62*(4), 1274–1284. https://doi.org/10.15517/rbt.v62i4.13231

Dickinson, J. L., Zuckerberg, B., & Bonter, D. N. (2010). Citizen Science as an Ecological Research Tool: Challenges and Benefits. *Annual Review of Ecology, Evolution, and Systematics*, *41*(1), 149–172. https://doi.org/10.1146/annurev-ecolsys-102209-144636

Dietz, H., Kueffer, C., & Parks, C. G. (2006). MIREN: A New Research Network Concerned With Plant Invasion into Mountain Areas. *Mountain Research and Development*, *26*(1), 80–81. https://doi.org/10.1659/0276-4741(2006)026[0080:MANRNC]2.0.CO;2

Diez, J. M., Williams, P. A., Randall, R. P., Sullivan, J. J., Hulme, P. E., & Duncan, R. P. (2009). Learning from failures: Testing broad taxonomic hypotheses about plant naturalization. *Ecology Letters*, *12*(11), 1174–1183. https://doi.org/10.1111/j.1461-0248.2009.01376.x

Donaldson, S., & Mudd, T. (2010). Sustaining cooperative weed management areas in the long-term. In E. Rindos (Ed.), *Plant Invasions: Policies, Politics, and Practices* (pp. 88–94). Center for Invasive Plant Management. https://files.cfc.umt.edu/cesu/FWS/2010/10Galli-Noble_Weeds_Across_Boarders%20_Proceedings.pdf

Doughty, R. W. (2000). *The Eucalyptus: A natural and commercial history of the gum tree.* Johns Hopkins University Press.

Driscoll, D. A., Catford, J. A., Barney, J. N., Hulme, P. E., Inderjit, Martin, T. G., Pauchard, A., Pyšek, P., Richardson, D. M., Riley, S., & Visser, V. (2014). New pasture plants intensify invasive species risk. *Proceedings of the National Academy of Sciences*, *111*(46), 16622–16627. https://doi.org/10.1073/pnas.1409347111

Duffy, G. A., Coetzee, B. W. T., Latombe, G., Akerman, A. H., McGeoch, M. A., & Chown, S. L. (2017). Barriers to globally invasive species are weakening across the Antarctic. *Diversity and Distributions*, *23*(9), 982–996. https://doi.org/10.1111/ddi.12593

Durant, S., & Faunce, T. A. (2018). Analysis of Australia's new biosecurity legislation. *Journal of Law and Medicine*, *25*(3), 647–654. https://pubmed.ncbi.nlm.nih.gov/29978659/

Dutta, W., Basuthakur, P., & Ray, P. (2021). Gardening the menace! *Environmental and Sustainability Indicators*, *12*, 100148. https://doi.org/10.1016/j.indic.2021.100148

Early, R., Bradley, B. A., Dukes, J. S., Lawler, J. J., Olden, J. D., Blumenthal, D. M., Gonzalez, P., Grosholz, E. D., Ibañez, I., Miller, L. P., Sorte, C. J. B., & Tatem, A. J. (2016). Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, *7*(1), 12485. https://doi.org/10.1038/ncomms12485

EASIN. (2021). *Guidelines and Codes of Conduct*. European Alien Species Information Network. https://easin.jrc.ec.europa.eu/easin/Documentation/Codesofconduct

EDDMapS. (2019). *EDDMapS*. University of Georgia – Center for Invasive Species and Ecosystem Health. https://www.eddmaps.org/index.cfm

Egoh, B. N., Ntshotsho, P., Maoela, M. A., Blanchard, R., Ayompe, L. M., & Rahlao, S. (2020). Setting the scene for achievable post-2020 convention on biological diversity targets: A review of the impacts of invasive alien species on ecosystem services in Africa. *Journal of Environmental Management*, *261*, 110171. https://doi.org/10.1016/j.jenvman.2020.110171

Ekanayake, E. M. B. P., Xie, Y., Ibrahim, A. S., Karunaratne, N. T. P., & Ahmad, S. (2020). Effective governance for management of invasive alien plants: Evidence from the perspective of forest and wildlife officers in Sri Lanka. *PeerJ*, *8*(1), e8343. https://doi.org/10.7717/peerj.8343

Eldridge, K., Davidson, J., Harwood, C., & Wyk, G. van. (1994). *Eucalypt domestication and breeding*. Clarendon Press.

Emery, S. B., Mulder, H. A. J., & Frewer, L. J. (2015). Maximizing the Policy Impacts of Public Engagement: A European Study. *Science Technology and Human Values*, *40*(3), 421–444. https://doi.org/10.1177/0162243914550319

Engelen, A. H., Serebryakova, A., Ang, P., Britton-Simmons, K., Mineur, F., Pedersen, M. F., Arenas, F., Fernández, C., Steen, H., Svenson, R., Pavia, H., Toth, G., Viard, F., & Santos, R. (2015). Circumglobal Invasion by the Brown Seaweed *Sargassum muticum*. In *Oceanography and Marine Biology* (Vol. 53, pp. 81–126). CRC Press. https://doi.org/10.1201/b18733-4

Epanchin-Niell, R. S. (2017). Economics of invasive species policy and management. In *Biological Invasions* (Vol. 19, Issue 11, pp. 3333–3354). https://doi.org/10.1007/s10530-017-1406-4

Epanchin-Niell, R. S., Hufford, M. B., Aslan, C. E., Sexton, J. P., Port, J. D., & Waring, T. M. (2010). Controlling invasive species in complex social landscapes. *Frontiers in Ecology and the Environment*, *8*(4), 210–216. https://doi.org/10.1890/090029

EPPO. (2009). PM 3/74(1): EPPO guidelines on the development of a Code of conduct on horticulture and invasive alien plants. *EPPO Bulletin*, *39*(3), 263–266. https://doi.org/10.1111/j.1365-2338.2009.02306.x

Esler, K. J., Prozesky, H., Sharma, G. P., & McGeoch, M. A. (2010). How wide is the "knowing-doing" gap in invasion biology? *Biological Invasions*, *12*(12), 4065–4075. https://doi.org/10.1007/s10530-010-9812-x

Essl, F., Bacher, S., Genovesi, P., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kowarik, I., Kühn, I., Pyšek, P., Rabitsch, W., Schindler, S., van Kleunen, M., Vilà, M., Wilson, J. R. U., & Richardson, D. M. (2018). Which taxa are alien? Criteria, applications, and uncertainties. *BioScience*, *68*(7), 496–509. https://doi.org/10.1093/biosci/biy057

Essl, F., Dullinger, S., Rabitsch, W., Hulme, P. E., Hülber, K., Jarošík, V., Kleinbauer, I., Krausmann, F., Kühn, I., Nentwig, W., Vilà, M., Genovesi, P., Gherardi, F., Desprez-Loustau, M.-L., Roques, A., &

Pyšek, P. (2011). Socioeconomic legacy yields an invasion debt. *Proceedings of the National Academy of Sciences*, *108*(1), 203–207. https://doi.org/10.1073/pnas.1011728108

Essl, F., Latombe, G., Lenzner, B., Pagad, S., Seebens, H., Smith, K., Wilson, J. R. U., & Genovesi, P. (2020). The Convention on Biological Diversity (CBD)'s Post-2020 target on invasive alien species – what should it include and how should it be monitored? *NeoBiota*, *62*, 99–121. https://doi.org/10.3897/neobiota.62.53972

Essl, F., Lenzner, B., Bacher, S., Bailey, S., Capinha, C., Daehler, C., Dullinger, S., Genovesi, P., Hui, C., Hulme, P. E., Jeschke, J. M., Katsanevakis, S., Kühn, I., Leung, B., Liebhold, A., Liu, C., MacIsaac, H. J., Meyerson, L. A., Nuñez, M. A., … Roura-Pascual, N. (2020). Drivers of future alien species impacts: An expert-based assessment. *Global Change Biology*, *26*(9), 4880–4893. https://doi.org/10.1111/gcb.15199

Estévez, R. A., Anderson, C. B., Pizarro, J. C., & Burgman, M. A. (2015). Clarifying values, risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species management. *Conservation Biology*, *29*(1), 19–30. https://doi.org/10.1111/cobi.12359

Estoup, A., & Guillemaud, T. (2010). Reconstructing routes of invasion using genetic data: Why, how and so what? *Molecular Ecology*, *19*(19), 4113–4130. https://doi.org/10.1111/j.1365-294X.2010.04773.x

EU. (2018). *Directive (EU) 2018/2001 of the European Parliament and of the Council of 11 December 2018 on the promotion of the use of energy from renewable sources (Text with EEA relevance.)*. Official Journal of the European Union. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=uriserv:OJ.L_.2018.328.01.0082.01.ENG&toc=OJ:L:2018:328:TOC

European Commission. (2021). *Reforming the WTO – Towards a sustainable and effective multilateral trading system*. Publications Office of the European Union. https://data.europa.eu/doi/10.2781/272480

European Environment Agency. (2010a). *Assessing biodiversity in Europe: The 2010 report*. Publications Office. https://data.europa.eu/doi/10.2800/42824

European Environment Agency. (2010b). *Towards an early warning and information system for invasive alien species (IAS) threatening biodiversity in Europe* (EEA Technical Report 5; p. 47).

Evans, J. M., Wilkie, A. C., & Burkhardt, J. (2008). Adaptive management of nonnative species: Moving beyond the "either-or" through experimental pluralism. *Journal of Agricultural and Environmental Ethics*, *21*(6), 521–539. https://doi.org/10.1007/s10806-008-9118-5

Evans, T., & Blackburn, T. M. (2020). Global variation in the availability of data on the environmental impacts of alien birds. *Biological Invasions*, *22*, 1027–1036. https://doi.org/10.1007/s10530-019-02153-z

Evans, T., Ermgassen, P. zu, Amano, T., & Peh, K. S.-H. (2018). Does governance play a role in the distribution of invasive alien species? *Ecology and Evolution*, *8*(4), 1984–1994. https://doi.org/10.1002/ece3.3744

Eysenbach, G. (2006). Citation Advantage of Open Access Articles. *PLoS Biology*, *4*(5), e157. https://doi.org/10.1371/journal.pbio.0040157

Failing, L., Gregory, R., & Higgins, P. (2013). Science, Uncertainty, and Values in Ecological Restoration: A Case Study in Structured Decision-Making and Adaptive Management. *Restoration Ecology*, *21*(4), 422–430. https://doi.org/10.1111/j.1526-100X.2012.00919.x

Fajardo del Castillo, T. (2016). The European Union's approach in the fight against wildlife trafficking: Challenges ahead. *Journal of International Wildlife Law & Policy*, *19*(1), 1–21. https://doi.org/10.1080/13880292.2016.1131516

Falk, B. G., Snow, R. W., & Reed, R. N. (2016). Prospects and Limitations of Citizen Science in Invasive Species Management: A Case Study with Burmese Pythons in Everglades National Park. *Southeastern Naturalist*, *15*(sp8), 89–102. https://doi.org/10.1656/058.015.sp806

Falkenmark, M. (2007). Good ecosystem governance: Balancing ecosystems and social needs. In A. R. Turton, H. J. Hattingh, G. A. Maree, D. J. Roux, M. Claassen, & W. F. Strydom (Eds.), *Governance as a Trialogue: Government-Society-Science in Transition* (pp. 59–76). Springer. https://doi.org/10.1007/978-3-540-46266-8_4

FAO. (2003). *Report of the Fifth Interim Commission on Phytosanitary Measures, 7-11 April 2003.* (ICPM 03). Food and Agriculture Organization of the United Nations. https://assets.ippc.int/static/media/files/publication/en/2016/11/Report_ICPM-5_2003_Apr_2003-05-23_BneT8OP.pdf

FAO. (2011). *A value chain approach to animal diseases risk management – Technical foundations and practical framework for field application* (N. Taylor & J. Rushton, Eds.). Food and Agriculture Organization of the United Nations.

FAO. (2016). *National Aquaculture Sector Overview–Argentina*. National aquaculture sector overview fact sheets. Food and Agriculture …. https://www.fao.org/fishery/en/countrysector/naso_argentina

FAO. (2019). *The State of the World's Biodiversity for Food and Agriculture* (D. Pilling & J. Bélanger, Eds.). Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/CA3129EN

Faulkner, K. T., Hulme, P. E., Pagad, S., Wilson, J. R. U., & Robertson, M. P. (2020). Classifying the introduction pathways of alien species: Are we moving in the right direction? *NeoBiota*, *62*, 143–159. https://doi.org/10.3897/neobiota.62.53543

Faulkner, K. T., Robertson, M. P., Rouget, M., & Wilson, J. R. U. (2016). Understanding and managing the introduction pathways of alien taxa: South Africa as a case study. *Biological Invasions*, *18*(1), Article 1. https://doi.org/10.1007/s10530-015-0990-4

Fenichel, E. P., & Abbott, J. K. (2014). Natural capital: From metaphor to measurement. *Journal of the Association of Environmental and Resource Economists*, *1*(1/2), 1–27. https://doi.org/10.1086/6760341

Fenichel, E. P., Richards, T. J., & Shanafelt, D. W. (2014). The Control of Invasive Species on Private Property with Neighbor-to-Neighbor Spillovers. *Environmental and Resource Economics*, *59*(2), 231–255. https://doi.org/10.1007/s10640-013-9726-z

Féral, J.-P. (2002). How useful are the genetic markers in attempts to understand and manage marine biodiversity? *Journal of Experimental Marine Biology and Ecology*, *268*(2), 121–145. https://doi.org/10.1016/S0022-0981(01)00382-3

Fernandez, L. (2011). Economic incentives to prevent aquatic invasive species in wetlands. *Canadian Journal of Agricultural Economics/Revue Canadienne d'agroeconomie*, *59*(2), 281–293. https://doi.org/10.1111/j.1744-7976.2011.01218.x

Finkel, S. E., & Muller, E. N. (1998). Rational Choice and the Dynamics of Collective Political Action: Evaluating Alternative Models with Panel Data. *The American Political Science Review*, *92*(1), 37–49. https://doi.org/10.2307/2585927

Fischer, A., Selge, S., van der Wal, R., & Larson, B. M. H. (2014). The Public and Professionals Reason Similarly about the Management of Non-Native Invasive Species: A Quantitative Investigation of the Relationship between Beliefs and Attitudes. *PLoS ONE*, *9*(8), e105495. https://doi.org/10.1371/journal.pone.0105495

Ford-Thompson, A. E. S., Snell, C., Saunders, G., & White, P. C. L. (2012). Stakeholder Participation in Management of Invasive Vertebrates. *Conservation Biology*, *26*(2), 345–356. https://doi.org/10.1111/j.1523-1739.2011.01819.x

Fournier, A., Penone, C., Grazia Pennino, M., & Courchamp, F. (2019). Predicting future invaders and future invasions. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(16), 7905–7910. https://doi.org/10.1073/pnas.1803456116

Foxcroft, L. C., Pyšek, P., Richardson, D. M., & Genovesi, P. (Eds.). (2013). *Plant invasions in protected areas: Patterns, problems and challenges* (Vol. 7). Springer Science & Business Media. https://doi.org/10.1007/978-94-007-7750-7

Foxcroft, L. C., Pyšek, P., Richardson, D. M., Genovesi, P., & MacFadyen, S. (2017). Plant invasion science in protected areas: Progress and priorities. *Biological Invasions*, *19*(5), 1353–1378. https://doi.org/10.1007/s10530-016-1367-z

Francour, P., Mangialajo, L., & Pastor, J. (2010). Mediterranean marine protected areas and non-indigenous fish spreading. In D. Golani & B. Appelbaum-Golani (Eds.), *Fish Invasions of the Mediterranean Sea: Change and Renewal* (pp. 127–144). Pensoft Publishers.

Frankham, R., Ballou, J. D., & Briscoe, D. A. (2010). *Introduction to Conservation Genetics*. Cambridge University Press. https://doi.org/10.1017/CBO9780511808999

Franklin, J., Regan, H. M., Hierl, L. A., Deutschman, D. H., Johnson, B. S., & Winchell, C. S. (2011). Planning, implementing, and monitoring multiple-species habitat conservation plans. *American Journal of Botany*, *98*(3), 559–571. http://www.jstor.org/stable/41149205

Frenot, Y., Chown, S. L., Whinam, J., Selkirk, P. M., Convey, P., Skotnicki, M., & Bergstrom, D. M. (2005). Biological invasions in the Antarctic: Extent, impacts and implications. *Biological Reviews*, *80*(1), 45–72. https://doi.org/10.1017/S1464793104006542

Gaertner, M., Biggs, R., Te Beest, M., Hui, C., Molofsky, J., & Richardson, D. M. (2014). Invasive plants as drivers of regime shifts: Identifying high-priority invaders that alter feedback relationships. *Diversity and Distributions*, *20*(7), 733–744. https://doi.org/10.1111/ddi.12182

Gaertner, M., Wilson, J. R. U., Cadotte, M. W., MacIvor, J. S., Zenni, R. D., & Richardson, D. M. (2017). Non-native species in urban environments: Patterns, processes, impacts and challenges. *Biological Invasions*, *19*(12), 3461–3469. https://doi.org/10.1007/s10530-017-1598-7

Galil, B. S., Marchini, A., Occhipinti-Ambrogi, A., Minchin, D., Narščius, A., Ojaveer, H., & Olenin, S. (2014). International arrivals: Widespread bioinvasions in European Seas. *Ethology Ecology & Evolution*, *26*(2–3), 152–171. https://doi.org/10.1080/03949370.2014.897651

Galil, B. S., Marchini, A., Occhipinti-Ambrogi, A., & Ojaveer, H. (2017). The enlargement of the Suez Canal – Erythraean introductions and management challenges. *Management of Biological Invasions*, *8*(2), 141–152. https://doi.org/10.3391/mbi.2017.8.2.02

Gallardo, B., Aldridge, D. C., González-Moreno, P., Pergl, J., Pizarro, M., Pyšek, P., Thuiller, W., Yesson, C., & Vilà, M. (2017). Protected areas offer refuge from invasive species spreading under climate change. *Global Change Biology*, *23*(12), 5331–5343. https://doi.org/10.1111/gcb.13798

García-Díaz, P., Ross, J. V., Woolnough, A. P., & Cassey, P. (2017). The illegal wildlife trade is a likely source of alien species. *Conservation Letters*, *10*(6), 690–698. https://doi.org/10.1111/conl.12301

García-Llorente, M., Martín-López, B., González, J. A., Alcorlo, P., & Montes, C. (2008). Social perceptions of the impacts and benefits of invasive alien species: Implications for management. *Biological Conservation*, *141*(12), 2969–2983. https://doi.org/10.1016/j.biocon.2008.09.003

GBIF Secretariat & IAIA. (2020). *Best Practices for Publishing Biodiversity Data from Environmental Impact Assessments. Version 1.0.* Global Biodiversity Information Facility, International Association for Impact Assessment. https://doi.org/10.35035/doc-5xdm-8762

GEF. (2016). *Proyecto ENEEI* [Global Environment Facility]. Argentina.gob.ar. https://www.argentina.gob.ar/ambiente/biodiversidad/exoticas-invasoras/proyecto

Geng, Y., Sarkis, J., & Bleischwitz, R. (2019). How to globalize the circular economy. *Nature*, *565*(7738), 153–155. https://doi.org/10.1038/d41586-019-00017-z

Genovesi, P. (2005). Eradications of invasive alien species in Europe: A review. *Biological Invasions*, *7*, 133. https://doi.org/10.1007/s10530-004-9642-9

Genovesi, P., Carboneras, C., Vilà, M., & Walton, P. (2015). EU adopts innovative legislation on invasive species: A step towards a global response to biological invasions? *Biological Invasions*, *17*(5), 1307–1311. https://doi.org/10.1007/s10530-014-0817-8

Genovesi, P., & Monaco, A. (2013). Guidelines for addressing invasive species in protected areas. In L. C. Foxcroft, P. Pyšek, D. M. Richardson, & P. Genovesi (Eds.), *Plant invasions in protected areas: Patterns, problems and challenges* (Vol. 7, pp. 487–506). Springer. https://doi.org/10.1007/978-94-007-7750-7_22

Genovesi, P., & Shine, C. (2004). *European strategy on invasive alien species: Convention on the Conservation of European Wildlife and Habitats (Bern Convention)* (Nature and Environment 137). Council of Europe. https://www.cbd.int/doc/external/cop-09/bern-01-en.pdf

Gerrits, L., & Edelenbos, J. (2004). Management of sediments through stakeholder involvement—The risks and value of engaging stakeholders when looking for solutions for sediment-related problems. *Journal of Soils and Sediments*, *4*(4), 239–246. https://doi.org/10.1065/jss2004.10.121

Giakoumi, S., Katsanevakis, S., Albano, P. G., Azzurro, E., Cardoso, A. C., Cebrian, E., Deidun, A., Edelist, D., Francour, P., Jimenez, C., Mačić, V., Occhipinti-Ambrogi, A., Rilov, G., & Sghaier, Y. R. (2019). Management priorities for marine invasive species. *Science of The Total Environment*, *688*, 976–982. https://doi.org/10.1016/j.scitotenv.2019.06.282

Giakoumi, S., Katsanevakis, S., Vassilopoulou, V., Panayotidis, P., Kavadas, S., Issaris, Y., Kokkali, A., Frantzis, A., Panou, A., & Mavrommati, G. (2012). Could European marine conservation policy benefit from systematic conservation planning? *Aquatic Conservation-Marine and Freshwater Ecosystems*, *22*(6), 762–775. https://doi.org/10.1002/aqc.2273

Giakoumi, S., Pey, A., Di Franco, A., Francour, P., Kizilkaya, Z., Arda, Y., Raybaud, V., & Guidetti, P. (2019). Exploring the relationships between marine protected areas and invasive fish in the world's most invaded sea. *Ecological Applications: A Publication of the Ecological Society of America*, *29*(1), e01809. https://doi.org/10.1002/eap.1809

Gilek, M., & Karlsson, M. (2016). Seeking Pathways Towards Improved Environmental Governance of the Baltic Sea. In M. Gilek, M. Karlsson, S. Linke, & K. Smolarz (Eds.), *Environmental Governance of the Baltic Sea* (Vol. 10, pp. 229–246). Springer. https://doi.org/10.1007/978-3-319-27006-7_10

Gilek, M., Karlsson, M., Linke, S., & Smolarz, K. (2016). Environmental Governance of the Baltic Sea: Identifying Key Challenges, Research Topics and Analytical Approaches. In M. Gilek, M. Karlsson, S. Linke, & K. Smolarz (Eds.), *Environmental Governance of the Baltic Sea* (Vol. 10, pp. 1–17). Springer. https://doi.org/10.1007/978-3-319-27006-7_1

Gilna, B., Kuzma, J., & Otts, S. S. (2014). Governance of genetic biocontrol technologies for invasive fish. *Biological Invasions*, *16*(6), 1299–1312. https://doi.org/10.1007/s10530-012-0367-x

Giltrap, N., Eyre, D., & Reed, P. (2009). Internet sales of plants for planting–an increasing trend and threat? *EPPO Bulletin*, *39*(2), 168–170. https://doi.org/10.1111/j.1365-2338.2009.02283.x

Giovos, I., Kleitou, P., Poursanidis, D., Batjakas, I., Bernardi, G., Crocetta, F., Doumpas, N., Kalogirou, S., Kampouris, T. E., Keramidas, I., Langeneck, J., Maximiadi, M., Mitsou, E., Stoilas, V.-O., Tiralongo, F., Romanidis-Kyriakidis, G., Xentidis, N.-J., Zenetos, A., & Katsanevakis, S. (2019). Citizen-science for monitoring marine invasions and stimulating public engagement: A case project from the eastern Mediterranean. *Biological Invasions*, *21*(12), 3707–3721. https://doi.org/10.1007/s10530-019-02083-w

Gippet, J. M. W., & Bertelsmeier, C. (2021). Invasiveness is linked to greater commercial success in the global pet trade. *Proceedings of the National Academy of Sciences*, *118*(14), e2016337118. https://doi.org/10.1073/pnas.2016337118

Goble, C. A., Bhagat, J., Aleksejevs, S., Cruickshank, D., Michaelides, D., Newman, D., Borkum, M., Bechhofer, S., Roos, M., Li, P., & De Roure, D. (2010). myExperiment: A repository and social network for the sharing of bioinformatics workflows. *Nucleic Acids Research*, *38*(suppl_2), W677–W682. https://doi.org/10.1093/nar/gkq429

Godfree, R., Firn, J., Johnson, S., Knerr, N., Stol, J., & Doerr, V. (2017). Why non-native grasses pose a critical emerging threat to biodiversity conservation, habitat connectivity and agricultural production in multifunctional rural landscapes. *Landscape Ecology*, *32*, 1219–1242. https://doi.org/10.1007/s10980-017-0516-9

Gómez Lozano, R., Anderson, L., Akins, J. L., Buddo, D. St. A., García-Moliner, G., Lozano, R., Gourdin, F., Laurent, M., Lilyestrom, C., Morris, J. A., Ramnanan, N., & Torres, R. (2013). *Regional strategy for the control of invasive lionfish in the Wider Caribbean.* (p. 31). International Coral Reef Initiative. https://www.cabdirect.org/cabdirect/abstract/20178100072

Goss, E. M., Tabima, J. F., Cooke, D. E., Restrepo, S., Fry, W. E., Forbes, G. A., Fieland, V. J., Cardenas, M., & Grünwald, N. J. (2014). The Irish potato famine pathogen *Phytophthora infestans* originated in central Mexico rather than the Andes. *Proceedings of the National Academy of Sciences*, *111*(24), 8791–8796. https://doi.org/10.1073/pnas.1401884111

Graham, S. (2013). Three cooperative pathways to solving a collective weed management problem. *Australasian Journal of Environmental Management*, *20*(2), 116–129. https://doi.org/10.1080/14486563.2013.774681

Graham, S. (2014). A new perspective on the trust power nexus from rural Australia. *Journal of Rural Studies*, *36*, 87–98. https://doi.org/10.1016/j.jrurstud.2014.06.010

Graham, S. (2019). Coordinating invasive plant management among conservation and rural stakeholders. *Land Use Policy*, *81*, 247–255. https://doi.org/10.1016/j.landusepol.2018.10.043

Graham, S., Metcalf, A. L., Gill, N., Niemiec, R., Moreno, C., Bach, T., Ikutegbe, V., Hallstrom, L., Ma, Z., & Lubeck, A. (2019). Opportunities for better use of collective action theory in research and governance for invasive species management. *Conservation Biology*, *33*(2), 275–287. https://doi.org/10.1111/cobi.13266

Graham, S., & Rogers, S. (2017). How Local Landholder Groups Collectively Manage Weeds in South-Eastern Australia. *Environmental Management*, *60*(3), 396–408. https://doi.org/10.1007/s00267-017-0859-7

Green, S. J., Akins, J. L., Maljković, A., & Côté, I. M. (2012). Invasive Lionfish Drive Atlantic Coral Reef Fish Declines. *PLoS ONE*, *7*(3), e32596. https://doi.org/10.1371/journal.pone.0032596

Groom, Q. J., Adriaens, T., Desmet, P., Simpson, A., De Wever, A., Bazos, I., Cardoso, A. C., Charles, L., Christopoulou, A., Gazda, A., Helmisaari, H., Hobern, D., Josefsson, M., Lucy, F., Marisavljevic, D., Oszako, T., Pergl, J., Petrovic-Obradovic, O., Prévot, C., … Vanderhoeven, S. (2017). Seven Recommendations to Make Your Invasive Alien Species Data More Useful. *Frontiers in Applied Mathematics and Statistics*, *3*, 13. https://doi.org/10.3389/fams.2017.00013

Groom, Q. J., Desmet, P., Reyserhove, L., Adriaens, T., Oldoni, D., Vanderhoeven, S., Baskauf, S. J., Chapman, A., McGeoch, M. A., Walls, R., Wieczorek, J., Wilson, J. R. U., Zermoglio, P. F., & Simpson, A. (2019). Improving Darwin Core for research and management of alien species. *Biodiversity Information Science and Standards*, *3*, e38084. https://doi.org/10.3897/biss.3.38084

Groom, Q. J., Desmet, P., Vanderhoeven, S., & Adriaens, T. (2015). The importance of open data for invasive alien species research, policy and management. *Management of Biological Invasions*, *6*(2), 119–125. https://doi.org/10.3391/mbi.2015.6.2.02

Guerrero, A. M., McAllister, R. R. J., & Wilson, K. A. (2015). Achieving Cross-Scale Collaboration for Large Scale Conservation Initiatives. *Conservation Letters*, *8*(2), 107–117. https://doi.org/10.1111/conl.12112

Guidetti, P., Baiata, P., Ballesteros, E., Di Franco, A., Hereu, B., Macpherson, E., Micheli, F., Pais, A., Panzalis, P., Rosenberg, A. A., Zabala, M., & Sala, E. (2014). Large-scale assessment of mediterranean marine protected areas effects on fish assemblages. *PLoS ONE*, *9*(4), e91841. https://doi.org/10.1371/journal.pone.0091841

Guneratne, A. (2002). *Many tongues, one people: The making of Tharu identity in Nepal*. Cornell University Press. https://www.jstor.org/stable/10.7591/j.ctv3s8mfn

Hackerott, S., Valdivia, A., Green, S. J., Côté, I. M., Cox, C. E., Akins, L., Layman, C. A., Precht, W. F., & Bruno, J. F. (2013). Native predators do not influence invasion success of Pacific lionfish on Caribbean reefs. *Plos One*, *8*(7), e68259. https://doi.org/10.1371/journal.pone.0068259

Halford, M., Heemers, L., Van Wesemael, D., Mathys, C., Wallens, S., Branquart, E., Vanderhoeven, S., Monty, A., & Mahy, G. (2014). The voluntary Code of conduct on invasive alien plants in Belgium: Results and lessons learned from the AlterIAS LIFE+ project. *EPPO Bulletin*, *44*(2), 212–222. https://doi.org/10.1111/epp.12111

Handerson, I. (2009). Progress of the UK Ruddy Duck eradication programme. *British Birds*, *102*(12), 680-690. https://britishbirds.co.uk/content/progress-uk-ruddy-duck-eradication-programme-0

Handford, C. E., Elliott, C. T., & Campbell, K. (2015). A review of the global pesticide legislation and the scale of challenge in reaching the global harmonization of food safety standards. *Integrated Environmental Assessment and Management*, *11*(4), 525–536. https://doi.org/10.1002/ieam.1635

Handl, G. (1998). The legal mandate of multilateral development banks as agents for change toward sustainable development. *American Journal of International Law*, *92*(4), 642–665. https://doi.org/10.2307/2998127

Hanley, N., & Roberts, M. (2019). The economic benefits of invasive species management. *People and Nature*, *1*(2), 124–137. https://doi.org/10.1002/pan3.31

Haque, Md. M., Beaumont, L. J., & Nipperess, D. A. (2020). Taxonomic shortfalls in digitised collections of Australia's flora. *Biodiversity and Conservation*, *29*(1), 333–343. https://doi.org/10.1007/s10531-019-01885-7

Hardisty, A. R., Belbin, L., Hobern, D., McGeoch, M. A., Pirzl, R., Williams, K. J., & Kissling, W. D. (2019). Research infrastructure challenges in preparing essential biodiversity variables data products for alien invasive species. *Environmental Research Letters*, *14*(2), 025005. https://doi.org/10.1088/1748-9326/aaf5db

Harvey, C. A., Dickson, B., & Kormos, C. (2010). Opportunities for achieving biodiversity conservation through REDD. *Conservation Letters*, *3*(1), 53–61. https://doi.org/10.1111/j.1755-263X.2009.00086.x

Head, L., & Atchison, J. (2015). Governing invasive plants: Policy and practice in managing the Gamba grass (*Andropogon gayanus*) – Bushfire nexus in northern Australia. *Land Use Policy*, *47*, 225–234. https://doi.org/10.1016/j.landusepol.2015.04.009

Herms, D. A., & McCullough, D. G. (2011). Pesticides for insect eradication. In D. Simberloff & M. Rejmanek (Eds.), *Encyclopedia of Biological Invasions* (Vol. 3, pp. 528–535). University of California Press. https://doi.org/10.1525/9780520948433-118

Herrick, C. N. (2019). A review of the U.S. invasive species policy mix: Questioning the prospect of an integrated regime. *Environmental Policy and Governance*, *29*(4), 262–278. https://doi.org/10.1002/eet.1852

Hershdorfer, M. E., Fernandez-Gimenez, M. E., & Howery, L. D. (2007). Key attributes influence the performance of local weed management programs in the southwest United States. *Rangeland Ecology and Management*, *60*(3), 225–234. https://doi.org/10.2111/1551-5028(2007)60[225:KAITPO]2.0.CO;2

Hewitt, C. L., & Campbell, M. L. (2007). Mechanisms for the prevention of marine bioinvasions for better biosecurity. *Marine Pollution Bulletin*, *55*(7–9), 395–401. https://doi.org/10.1016/j.marpolbul.2007.01.005

Hewitt, C. L., Campbell, M. L., & Gollasch, S. (2006). *Alien species in aquaculture: Considerations for responsible use*. IUCN. https://portals.iucn.org/library/sites/library/files/documents/2006-036.pdf

Hewitt, C. L., Willing, J., Bauckham, A., Cassidy, A. M., Cox, C. M. S., Jones, L., & Wotton, D. M. (2004). New Zealand marine biosecurity: Delivering outcomes in a fluid environment. *New Zealand Journal of Marine and Freshwater Research*, *38*(3), 429–438. https://doi.org/10.1080/00288330.2004.9517250

Heywood, V. H., & Brunel, S. (2009). *Code of conduct on horticulture and invasive alien plants*. Council of Europe.

Heywood, V. H., & Sharrock, S. (2013). *European code of conduct for botanic gardens on invasive alien species*. Council of Europe.

Hickerson, E. L., Schmahl, G. P., Johnston, M. A., Nuttall, M. F., Embesi, J. A., & Eckert, R. J. (2012, July 9). Flower Garden Banks–a refuge in the Gulf of Mexico? *Proceedings of the 12th International Coral Reef Symposium*. the 12th International Coral Reef Symposium, Cairns, Australia.

Hiller, T., & Haelewaters, D. (2019). A case of silent invasion: Citizen science confirms the presence of *Harmonia axyridis* (Coleoptera, Coccinellidae) in Central America. *PLoS ONE*, *14*(7), e0220082. https://doi.org/10.1371/journal.pone.0220082

Hoagstrom, C. W., Hayer, C.-A., & Berry, C. R. (2009). Criteria for determining native distributions of biota: The case of the northern plains killifish in the Cheyenne River drainage, North America. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *19*(1), 88–95. https://doi.org/10.1002/aqc.1000

Hoff, M. H., Irons, K. S., & Forester, J. E. (2019). Policy Challenges Relating to Integrated Pest Management of Freshwater Aquatic Invasive Animals. *North American Journal of Fisheries Management*, *41*(2), 276–288. https://doi.org/10.1002/nafm.10329

Holman, L. E., de Bruyn, M., Creer, S., Carvalho, G., Robidart, J., & Rius, M. (2019). Detection of introduced and resident marine species using environmental DNA metabarcoding of sediment and water. *Scientific Reports*, *9*(1), 11559. https://doi.org/10.1038/s41598-019-47899-7

Hortal, J., De Bello, F., Diniz-Filho, J. A. F., Lewinsohn, T. M., Lobo, J. M., & Ladle, R. J. (2015). Seven Shortfalls that Beset Large-Scale Knowledge of Biodiversity. *Annual Review of Ecology, Evolution, and Systematics*, *46*(1), 523–549. https://doi.org/10.1146/annurev-ecolsys-112414-054400

Howard, P. L. (2019). Human adaptation to invasive species: A conceptual framework based on a case study metasynthesis. *Ambio*, *48*(12), 1401–1430. https://doi.org/10.1007/s13280-019-01297-5

Howard, P. L., & Pecl, G. T. (2019). Introduction: Autochthonous human adaptation to biodiversity change in the Anthropocene. *Ambio*, *48*(12), 1389–1400. https://doi.org/10.1007/s13280-019-01283-x

Howard, T. M., Thompson, L. J., Frumento, P., & Alter, T. (2018). Wild dog management in Australia: An interactional approach to case studies of community-led action. *Human Dimensions of Wildlife*, *23*(3), 242–256. https://doi.org/10.1080/10871209.2017.1414337

Hughes, K. A., & Convey, P. (2010). The protection of Antarctic terrestrial ecosystems from inter- and intra-continental transfer of non-indigenous species by human activities: A review of current systems and practices. *Global Environmental Change*, *20*(1), 96–112. https://doi.org/10.1016/j.gloenvcha.2009.09.005

Hughes, K. A., & Convey, P. (2012). Determining the native/non-native status of newly discovered terrestrial and freshwater species in Antarctica—Current knowledge, methodology and management action. *Journal of Environmental Management*, *93*(1), 52–66. https://doi.org/10.1016/j.jenvman.2011.08.017

Hughes, K. A., & Convey, P. (2014). Alien invasions in Antarctica-is anyone liable? *Polar Research*, *33*(1), 22103. https://doi.org/10.3402/polar.v33.22103

Hughes, K. A., & Pertierra, L. R. (2016). Evaluation of non-native species policy development and implementation within the Antarctic Treaty area. *Biological Conservation*, *200*, 149–159. https://doi.org/10.1016/j.biocon.2016.03.011

Hughes, K. A., Pertierra, L. R., Molina-Montenegro, M. A., & Convey, P. (2015). Biological invasions in terrestrial Antarctica: What is the current status and can we respond? *Biodiversity and Conservation*, *24*(5), 1031–1055. https://doi.org/10.1007/s10531-015-0896-6

Hulata, G. (2001). Genetic manipulations in aquaculture: A review of stock improvement by classical and modern technologies. *Genetica*, *111*(1/3), 155–173. https://doi.org/10.1023/A:1013776931796

Hulme, P. E. (2009). Trade, transport and trouble: Managing invasive species pathways in an era of globalization. *Journal of Applied Ecology*, *46*(1), 10–18. https://doi.org/10.1111/j.1365-2664.2008.01600.x

Hulme, P. E. (2011). Biosecurity: The changing face of invasion biology. In D. M. Richardson (Ed.), *Fifty years of invasion ecology: The legacy of Charles Elton* (pp. 73–88). Wiley-Blackwell. https://doi.org/10.1002/9781444329988.ch23

Hulme, P. E. (2020). One Biosecurity: A unified concept to integrate human, animal, plant, and environmental health. *Emerging Topics in Life Sciences*, *4*(5), 539–549. https://doi.org/10.1042/ETLS20200067

Hulme, P. E. (2021). Advancing One Biosecurity to Address the Pandemic Risks of Biological Invasions. *BioScience*, *71*(7), 708–721. https://doi.org/10.1093/biosci/biab019

Hulme, P. E., Brundu, G., Carboni, M., Dehnen-Schmutz, K., Dullinger, S., Early, R., Essl, F., González-Moreno, P., Groom, Q. J., Kueffer, C., Kühn, I., Maurel, N., Novoa, A., Pergl, J., Pyšek, P., Seebens, H., Tanner, R., Touza, J. M., van Kleunen, M., & Verbrugge, L. N. H. (2018). Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions. *Journal of Applied Ecology*, *55*(1), 92–98. https://doi.org/10.1111/1365-2664.12953

IAATO. (2020). *Boot, clothing and equipment decontamination guidelines for small boat operators*. International Association of Antarctic Tour Operators. https://iaato.org/decontamination-guidelines

IFC. (2012). *Performance Standard 6: Biodiversity Conservation and Sustainable Management of Living Natural Resources*. International Finance Corporation. https://www.ifc.org/en/insights-reports/2012/ifc-performance-standard-6

IMO. (2004). *International Convention for the Control and Management of Ships' Ballast Water and Sediments (BMW)*. International Maritime Organization. https://www.imo.org/en/About/Conventions/Pages/International-Convention-for-the-Control-and-Management-of-Ships%27-Ballast-Water-and-Sediments-(BWM).aspx

IMO. (2022). *Status of multilateral conventions and instruments in respect of which the International Maritime Organization or its Secretary-General performs depositary or other functions as at 26 May 2022.* International Maritime Organization.

InforMEA. (2021). *Early warning system | InforMEA*. United Nations Information Portal on Multilateral Environmental Agreements. https://www.informea.org/en/terms/early-warning-system

IPBES. (2016). *The assessment report of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3402857

IPBES. (2018a). *Summary for policymakers of the assessment report on land degradation and restoration of the Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services.* Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237410

IPBES. (2018b). *Summary for policymakers of the regional assessment report on biodiversity and ecosystem services for Asia and the Pacific of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPBES. (2018c). *The IPBES regional assessment report on biodiversity and ecosystem services for Asia and the Pacific* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237374

IPBES. (2018d). *The IPBES regional assessment report on biodiversity and ecosystem services for Europe and Central Asia* (Report). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3237429

IPBES. (2019a). *Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3831673

IPBES. (2019b). Methodological guidance for assessing policy instruments and facilitating the use for policy support tools and methodologies through IPBES assessments. *Information on Work Related to Policy Support Tools and Methodologies*, *IPBES/7/INF/13*, Appendix 1.

IPBES. (2019c). *Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (summary for policy makers). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. https://doi.org/10.5281/ZENODO.3553458

IPBES. (2021). *Methodological guidance for recognizing and working with indigenous and local knowledge in IPBES*. https://ipbes.net/sites/default/files/inline-files/IPBES_ILK_MethGuide.pdf

IPBES. (2022). *Report of the third Indigenous and local knowledge dialogue workshop for the IPBES thematic assessment of invasive alien species and their control.* https://ipbes.net/sites/default/files/2023-02/IPBES_IAS_3rdILKDialogue_SPM-SOD_Report_FinalForWeb2.pdf

IPCC. (2007). *Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Intergovernmental Panel on Climate Change. https://www.semanticscholar.org/paper/Contribution-of-Working-Group-II-to-the-Fourth-of-Palut-Canziani/ec27546040387c2067df8278f28ad8fd17981eb6

IPPC. (1952). *International Plant Protection Convention*. International Plant Protection Convention. http://www.fao.org/fileadmin/user_upload/legal/docs/004s-e.pdf

IUCN. (2000). *IUCN guidelines for the prevention of biodiversity loss caused by alien invasive species*. IUCN Species Survival Commission (SSC), Invasive Species Specialist Group. https://portals.iucn.org/library/node/12413

IUCN. (2017). *Guidance for interpretation of CBD categories on introduction pathways* [Technical note prepared by IUCN for the European Commission]. International Union for Conservation of Nature. https://nora.nerc.ac.uk/id/eprint/519129/1/N519129CR.pdf

Jackson, S. M., Groves, C. P., Fleming, P. J. S., Aplin, K. P., Eldridge, M. D. B., Gonzalez, A., & Helgen, K. M. (2017). The Wayward Dog: Is the Australian native dog or Dingo a distinct species? *Zootaxa*, *4317*(2), 201. https://doi.org/10.11646/zootaxa.4317.2.1

Jacobs, K. R. (2017). Teams at Their Core: Implementing an "All LANDS Approach to Conservation" Requires Focusing on Relationships, Teamwork Process, and Communications. *Forests*, *8*(7), 246. https://doi.org/10.3390/f8070246

Jarić, I., Correia, R. A., Roberts, D. L., Gessner, J., Meinard, Y., & Courchamp, F. (2019). On the overlap between scientific and societal taxonomic attentions—Insights for conservation. *Science of The Total Environment*, *648*, 772–778. https://doi.org/10.1016/j.scitotenv.2018.08.198

Jerde, C. L., Mahon, A. R., Chadderton, W. L., & Lodge, D. M. (2011). "Sight-unseen" detection of rare aquatic species using environmental DNA. *Conservation Letters*, *4*(2), 150–157. https://doi.org/10.1111/j.1755-263X.2010.00158.x

Jeschke, J. M., Aparicio, L. G., Haider, S., Heger, T., Lortie, C. J., Pyšek, P., & Strayer, D. L. (2012). Taxonomic bias and lack of cross-taxonomic studies in invasion biology. *Frontiers in Ecology and the Environment*, *10*(7), 349–350. https://doi.org/10.1890/12.WB.016

Jeschke, J. M., Börner, K., Stodden, V., & Tockner, K. (2019). Open Access journals need to become first choice, in invasion ecology and beyond. *NeoBiota*, *52*, 1–8. https://doi.org/10.3897/neobiota.52.39542

Jetz, W., McGeoch, M. A., Guralnick, R., Ferrier, S., Beck, J., Costello, M. J., Fernandez, M., Geller, G. N., Keil, P., Merow, C., Meyer, C., Muller-Karger, F. E., Pereira, H. M., Regan, E. C., Schmeller, D. S., & Turak, E. (2019). Essential biodiversity variables for mapping and monitoring species populations. *Nature Ecology & Evolution*, *3*(4), Article 4. https://doi.org/10.1038/s41559-019-0826-1

Jhala, Y., Gopal, R., Mathur, V., Ghosh, P., Negi, H. S., Narain, S., Yadav, S. P., Malik, A., Garawad, R., & Qureshi, Q. (2021). Recovery of tigers in India: Critical introspection and potential lessons. *People and Nature*, *3*(2), 281–293. https://doi.org/10.1002/pan3.10177

Johnson, B. A., Mader, A. D., Dasgupta, R., & Kumar, P. (2020). Citizen science and invasive alien species: An analysis of citizen science initiatives using information and communications technology (ICT) to collect invasive alien species observations. *Global Ecology and Conservation*, *21*, e00812. https://doi.org/10.1016/j.gecco.2019.e00812

Johnston, M. A., Sterne, T. K., Eckert, R. J., Nuttall, M. F., Embesi, J. A., Walker, R. D., Hu, X., Hickerson, E. L., & Schmahl, G. P. (2017). *Long-term monitoring at East and West Flower Garden Banks: 2016 annual report* (ONMS-17-09; Marine Sanctuaries Conservation Series). U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary.

Jones, K. R., & Corona, J. P. (2008). An ambient tax approach to invasive species. In *Ecological Economics* (Vol. 64, Issue 3, pp. 534–541). https://doi.org/10.1016/j.ecolecon.2007.03.006

Jones, S. B., & Foote, L. E. (1991). *Gardening with native wild flowers*. Timber Press.

Jonsson, A. M., Bostrom, M., Dreyer, M., & Soderstrom, S. (2016). Risk Communication and the Role of the Public: Towards Inclusive Environmental Governance of the Baltic Sea? In M. Gilek, M. Karlsson, S. Linke, & K. Smolarz (Eds.), *Environmental Governance of the Baltic Sea* (Vol. 10, pp. 205–227). Springer. https://doi.org/10.1007/978-3-319-27006-7_9

Juffe-Bignoli, D., Brooks, T. M., Butchart, S. H., Jenkins, R. B., Boe, K., Hoffmann, M., Angulo, A., Bachman, S., Böhm, M., Brummitt, N., Carpenter, K. E., Comer, P. J., Cox, N., Cuttelod, A., Darwall, W. R. T., Marco, M. D., Fishpool, L. D. C., Goettsch, B., Heath, M., … Kingston, N. (2016). Assessing the cost of global biodiversity and conservation knowledge. *PLoS ONE*, *11*(8), e0160640. https://doi.org/10.1371/journal.pone.0160640

Justo-Hanani, R., & Dayan, T. (2021). Risk regulation and precaution in Europe and the United States: The case of bioinvasion. *Policy Sciences*, *54*, 3–20. https://doi.org/10.1007/s11077-020-09409-9

Kamelamela, K. L., Springer, H. K., Keakealani, R. K., Ching, M. U., Ticktin, T., Ohara, R. D., Parsons, E. W., Adkins, E. D., Francisco, K. S., & Giardina, C. (2022). *Kōkua aku, Kōkua mai*: An Indigenous Consensus-driven and Place-based Approach to Community Led Dryland Restoration and Stewardship. *Forest Ecology and Management*, *506*, 119949. https://doi.org/10.1016/j.foreco.2021.119949

Kamenova, S., Bartley, T. J., Bohan, D. A., Boutain, J. R., Colautti, R. I., Domaizon, I., Fontaine, C., Lemainque, A., Le Viol, I., Mollot, G., Perga, M.-E., Ravigné, V., & Massol, F. (2017). Chapter Three – Invasions Toolkit: Current Methods for Tracking the Spread and Impact of Invasive Species. In D. A. Bohan, A. J. Dumbrell, & F. Massol (Eds.), *Advances in Ecological Research* (Vol. 56, pp. 85–182). Academic Press. https://doi.org/10.1016/bs.aecr.2016.10.009

Katsanevakis, S., Genovesi, P., Gaiji, S., Hvid, H. N., Roy, H. E., Nunes, A. L., Aguado, F. S., Bogucarskis, K., Debusscher, B., Deriu, I., Harrower, C., Josefsson, M., Lucy, F., Marchini, A., Richards, G., Trichkova, T., Vanderhoeven, S., Zenetos, A., & Cardoso, A. C. (2013). Implementing the European policies for alien species – networking, science, and partnership in a complex environment. *Management of Biological Invasions*, *4*(1), 3–6. https://doi.org/10.3391/mbi.2013.4.1.02

Katsanevakis, S., & Roy, H. E. (2015). Alien species related information systems and information management. *Management of Biological Invasions*, *6*(2), 115–117. https://doi.org/10.3391/mbi.2015.6.2.01

Katsanevakis, S., Wallentinus, I., Zenetos, A., Leppäkoski, E., Çinar, M. E., Oztürk, B., Grabowski, M., Golani, D., & Cardoso, A. C. (2014). Impacts of invasive alien marine species on ecosystem services and biodiversity: A pan-European review. *Aquatic Invasions*, *9*(4), 391–423. https://doi.org/10.3391/ai.2014.9.4.01

Keller, R. P., & Lodge, D. M. (2007). Species invasions from commerce in live aquatic organisms: Problems and possible solutions. *Bioscience*, *57*(5), 428–436. https://doi.org/10.1641/B570509

Kirkfeldt, T. S. (2019). An ocean of concepts: Why choosing between ecosystem-based management, ecosystem-based approach and ecosystem approach makes a difference. *Marine Policy*, *106*, 103541. https://doi.org/10.1016/j.marpol.2019.103541

Kissling, W. D., Ahumada, J. A., Bowser, A., Fernandez, M., Fernández, N., García, E. A., Guralnick, R. P., Isaac, N. J. B., Kelling, S., Los, W., McRae, L., Mihoub, J.-B., Obst, M., Santamaria, M., Skidmore, A. K., Williams, K. J., Agosti, D., Amariles, D., Arvanitidis, C., … Hardisty, A. R. (2018). Building essential biodiversity variables (EBVs) of species distribution and abundance at a global scale. *Biological Reviews*, *93*(1), 600–625. https://doi.org/10.1111/brv.12359

Kissling, W. D., Hardisty, A., García, E. A., Santamaria, M., De Leo, F., Pesole, G., Freyhof, J., Manset, D., Wissel, S., Konijn, J., & Los, W. (2015). Towards global interoperability for supporting biodiversity research on essential biodiversity variables (EBVs). *Biodiversity*, *16*(2–3), 99–107. https://doi.org/10.1080/14888386.2015.1068709

Kleinschroth, F., Winton, R. S., Calamita, E., Niggemann, F., Botter, M., Wehrli, B., & Ghazoul, J. (2021). Living with floating vegetation invasions. *Ambio*, *50*(1), 125–137. https://doi.org/10.1007/s13280-020-01360-6

Kluger, L. C., Gorris, P., Kochalski, S., Mueller, M. S., & Romagnoni, G. (2020). Studying human–nature relationships through a network lens: A systematic review. *People and Nature*, *2*(4), 1100–1116. https://doi.org/10.1002/pan3.10136

Knowler, D., & Barbier, E. (2005). Importing exotic plants and the risk of invasion: Are market-based instruments adequate? *Ecological Economics*, *52*(3), 341–354. https://doi.org/10.1016/j.ecolecon.2004.06.019

Kocovsky, P. M., Sturtevant, R., & Schardt, J. (2018). What it is to be established: Policy and management implications for non-native and invasive species. *Management of Biological Invasions*, *9*(3), 177–185. https://doi.org/10.3391/mbi.2018.9.3.01

Kohl, P. A., Brossard, D., Scheufele, D. A., & Xenos, M. A. (2019). Public views about editing genes in wildlife for conservation. *Conservation Biology*, *33*(6), 1286–1295. https://doi.org/10.1111/cobi.13310

König, H. J., Kiffner, C., Kramer-Schadt, S., Fürst, C., Keuling, O., & Ford, A. T. (2020). Human–wildlife coexistence in a changing world. *Conservation Biology*, *34*(4), 786–794. https://doi.org/10.1111/cobi.13513

Koontz, T. M., & Thomas, C. W. (2006). What Do We Know and Need to Know about the Environmental Outcomes of Collaborative Management? *Public Administration Review*, *66*(s1), 111–121. https://doi.org/10.1111/j.1540-6210.2006.00671.x

Koop, A. L., Fowler, L., Newton, L. P., & Caton, B. P. (2012). Development and validation of a weed screening tool for the United States. *Biological Invasions*, *14*(2), 273–294. https://doi.org/10.1007/s10530-011-0061-4

Kotchen, M. J. (2013). Voluntary-and information-based approaches to environmental management: A public economics perspective. *Review of Environmental Economics and Policy*, *7*(2), 276–295. https://doi.org/10.1093/reep/ret012

Kowalski, T. (2006). *Chalara fraxinea* sp. Nov. Associated with dieback of ash (*Fraxinus excelsior*) in Poland. *Forest Pathology*, *36*(4), 264–270. https://doi.org/10.1111/j.1439-0329.2006.00453.x

Kramer, L. (2005). Discussions on Directive 2004/35 Concerning Environmental Liability. *Journal for European Environmental & Planning Law*, *2*(4), 250–256. https://doi.org/10.1163/187601005X00020

Kretser, H. E., Wong, R., Roberton, S., Pershyn, C., Huang, J., Sun, F., Kang, A., & Zahler, P. (2015). Mobile decision-tree tool technology as a means to detect wildlife crimes and build enforcement networks. *Biological Conservation*, *189*, 33–38. https://doi.org/10.1016/j.biocon.2014.08.018

Kridel, K. (2008). *State says goodbye to tree-killing pest*. Chicago Tribune. https://www.chicagotribune.com/news/ct-xpm-2008-04-18-0804170845-story.html

Kropf, B., Schmid, E., Schönhart, M., & Mitter, H. (2020). Exploring farmers' behavior toward individual and collective measures of Western Corn Rootworm control – A case study in south-east Austria. *Journal of Environmental Management*, *264*, 110431. https://doi.org/10.1016/j.jenvman.2020.110431

Kruger, H. P. (2016). Adaptive co-management for collaborative commercial pest management: The case of industry-driven fruit fly area-wide management. *International Journal of Pest Management*, *62*(4), 336–347. https://doi.org/10.1080/09670874.2016.1206992

Kubiszewski, I., Costanza, R., Franco, C., Lawn, P., Talberth, J., Jackson, T., & Aylmer, C. (2013). Beyond GDP: Measuring and achieving global genuine progress. *Ecological Economics*, *93*, 57–68. https://doi.org/10.1016/j.ecolecon.2013.04.019

Kudsk, P., & Mathiassen, S. K. (2020). Pesticide regulation in the European Union and the glyphosate controversy. *Weed Science*, *68*(3), 214–222. https://doi.org/10.1017/wsc.2019.59

Kueffer, C., Daehler, C., Dietz, H., McDougall, K., Parks, C., Pauchard, A., & Rew, L. (2014). The *Mountain Invasion Research Network (MIREN)*. Linking Local and Global Scales for Addressing an Ecological Consequence of Global Change. *GAIA – Ecological Perspectives for Science and Society*, *23*(3), 263–265. https://doi.org/10.14512/gaia.23.3.11

Lambin, X., Bryce, R., Fraser, E., J., Melero, Y., Oliver, M., K., & Walker, H. (2012). Turning Back the Tide of American Mink Invasion in Partnership with Communities. *Proceedings of the Vertebrate Pest Conference*, *25*(25), 28–33. https://doi.org/10.5070/V425110616

Lambin, X., Oliver, M., Bryce, R., Urquhart, J., Davies, L., & Gray, H. (2011). *Turning back the tide of American mink invasion in partnership with communities*.

Lane, M. B., & Corbett, T. (2005). The tyranny of localism: Indigenous participation in community-based environmental management. *Journal of Environmental Policy and Planning*, *7*(2), 141–159. https://doi.org/10.1080/15239080500338671

Lansink, A. O., Schut, M., Kamanda, J., & Klerkx, L. (2018). A multi-level and multi-actor approach to risk governance: A conceptual framework to support policy development for Ambrosia weed control. *Journal of Risk Research*, *21*(6), 780–799. https://doi.org/10.1080/13669877.2016.1247376

Larson, D. L., Phillips-Mao, L., Quiram, G., Sharpe, L., Stark, R., Sugita, S., & Weiler, A. (2011). A framework for sustainable invasive species management: Environmental, social, and economic objectives. *Journal of Environmental Management*, *92*(1), 14–22. https://doi.org/10.1016/j.jenvman.2010.08.025

Larson, E. R., Howell, S., Kareiva, P., & Armsworth, P. R. (2016). Constraints of philanthropy on determining the distribution of biodiversity conservation funding. *Conservation Biology*, *30*(1), 206–215. https://doi.org/10.1111/cobi.12608

Latombe, G., Pyšek, P., Jeschke, J. M., Blackburn, T. M., Bacher, S., Capinha, C., Costello, M. J., Fernández, M., Gregory, R. D., Hobern, D., Hui, C., Jetz, W., Kumschick, S., McGrannachan, C., Pergl, J., Roy, H. E., Scalera, R., Squires, Z. E., Wilson, J. R. U., … McGeoch, M. A. (2017). A vision for global monitoring of biological invasions. *Biological Conservation*, *213*, 295–308. https://doi.org/10.1016/j.biocon.2016.06.013

Le Roux, P. C., & McGeoch, M. A. (2008). Changes in climate extremes, variability and signature on sub-Antarctic Marion Island. *Climatic Change*, *86*, 309–329. https://doi.org/10.1007/s10584-007-9259-y

Leadley, P. W., Krug, C. B., Alkemade, R., Pereira, H. M., Sumaila, U. R., Walpole, M., Marques, A., Newbold, T., Teh, L. S. L., van Kolck, J., Bellard, C., Januchowski-Hartley, S. R., & Mumby, P. J. (2014). *Progress towards the Aichi biodiversity targets: An assessment of biodiversity trends, policy scenarios and key actions* (Global Biodiversity Outlook 4 (GBO-4) Technical Report 78; CBD Technical Series, p. 500). Secretariat of the Convention on Biological Diversity. http://www.deslibris.ca/ID/244310

Lebouvier, M., Laparie, M., Hullé, M., Marais, A., Cozic, Y., Lalouette, L., Vernon, P., Candresse, T., Frenot, Y., & Renault, D. (2011). The significance of the sub-Antarctic Kerguelen Islands for the assessment of the vulnerability of native communities to climate change, alien insect invasions and plant viruses. *Biological Invasions*, *13*, 1195–1208. https://doi.org/10.1007/s10530-011-9946-5

Leclère, D., Obersteiner, M., Barrett, M., Butchart, S. H. M., Chaudhary, A., De Palma, A., DeClerck, F. A. J., Di Marco, M., Doelman, J. C., Dürauer, M., Freeman, R., Harfoot, M., Hasegawa, T., Hellweg, S., Hilbers, J. P., Hill, S. L. L., Humpenöder, F., Jennings, N., Krisztin, T., … Young, L. (2020). Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature*, *585*(7826), 551–556. https://doi.org/10.1038/s41586-020-2705-y

Lee, J. E., & Chown, S. L. (2011). Quantification of intra-regional propagule movements in the Antarctic. *Antarctic Science*, *23*(4), 337–342. https://doi.org/10.1017/S0954102011000198

Lehtiniemi, M., Lehmann, A., Javidpour, J., & Myrberg, K. (2012). Spreading and physico-biological reproduction limitations of the invasive American comb jelly *Mnemiopsis leidyi* in the Baltic Sea. *Biological Invasions*, *14*(2), 341–354. https://doi.org/10.1007/s10530-011-0066-z

Lemos, M. C., Arnott, J. C., Ardoin, N. M., Baja, K., Bednarek, A. T., Dewulf, A., Fieseler, C., Goodrich, K. A., Jagannathan, K., Klenk, N., Mach, K. J., Meadow, A. M., Meyer, R., Moss, R., Nichols, L., Sjostrom, K. D., Stults, M., Turnhout, E., Vaughan, C., … Wyborn, C. (2018). To co-produce or not to co-produce. *Nature Sustainability*, *1*(12), 722–724. https://doi.org/10.1038/s41893-018-0191-0

Lenda, M., Skórka, P., Knops, J. M. H., Moroń, D., Sutherland, W. J., Kuszewska, K., & Woyciechowski, M. (2014). Effect of the Internet Commerce on Dispersal Modes of Invasive Alien Species. *PLoS ONE*, *9*(6), e99786. https://doi.org/10.1371/journal.pone.0099786

Lescrauwaet, A.-K., Vandepitte, L., Fockedey, N., De Pooter, D., Verleye, T., & Mees, J. (2015). Invasive Alien Species in Belgian marine waters: An information platform and checklist for science and policy support. *Management of Biological Invasions*, *6*(2), 209–213. https://doi.org/10.3391/mbi.2015.6.2.11

Leventon, J., Abson, D. J., & Lang, D. J. (2021). Leverage points for sustainability transformations: Nine guiding questions for sustainability science and practice. *Sustainability Science*, *16*(3), 721–726. https://doi.org/10.1007/s11625-021-00961-8

Liebhold, A. M., Campbell, F. T., Gordon, D. R., Guo, Q., Havill, N., Kinder, B., MacKenzie, R., Lance, D. R., Pearson, D. E., Sing, S. E., Warziniack, T., Venette, R. C., & Yemshanov, D. (2021). The Role of International Cooperation in Invasive Species Research. In T. M. Poland, T. Patel-Weynand, D. M. Finch, C. F. Miniat, D. C. Hayes, & V. M. Lopez (Eds.), *Invasive Species in Forests and Rangelands of the United States* (pp. 293–303). Springer. https://doi.org/10.1007/978-3-030-45367-1_13

Liedtke, R., Barros, A., Essl, F., Lembrechts, J. J., Wedegärtner, R. E. M., Pauchard, A., & Dullinger, S. (2020). Hiking trails as conduits for the spread of non-native species in mountain areas. *Biological Invasions*, *22*(3), 1121–1134. https://doi.org/10.1007/s10530-019-02165-9

Lien, A. M., Baldwin, E., & Franklin, K. (2021). Collective Action and Invasive Species Governance in Southern Arizona. *Rangeland Ecology and Management*, *74*, 151–164. https://doi.org/10.1016/j.rama.2020.10.004

Lindgreen, A., & Swaen, V. (2010). Corporate Social Responsibility. *International Journal of Management Reviews*, *12*(1), 1–7. https://doi.org/10.1111/j.1468-2370.2009.00277.x

Linke, S., Gilek, M., & Karlsson, M. (2016). Science-Policy Interfaces in Baltic Sea Environmental Governance: Towards Regional Cooperation and Management of Uncertainty? In M. Gilek, M. Karlsson, S. Linke, & K. Smolarz (Eds.), *Environmental Governance of the Baltic Sea* (Vol. 10, pp. 173–203). Springer. https://doi.org/10.1007/978-3-319-27006-7_8

Liu, C., Wolter, C., Xian, W., & Jeschke, J. M. (2020). Species distribution models have limited spatial transferability for invasive species. *Ecology Letters*, *23*(11), 1682–1692. https://doi.org/10.1111/ele.13577

Liu, J., Dou, Y., Batistella, M., Challies, E., Connor, T., Friis, C., Millington, J. D. A., Parish, E., Romulo, C. L., Bicudo Silva, R. F., Triezenberg, H., Yang, H., Zhao, Z., Zimmerer, K. S., Huettmann, F., Treglia, M. L., Basher, Z., Chung, M. G., Herzberger, A., … Sun, J. (2018). Spillover systems in a telecoupled Anthropocene: Typology, methods, and governance for global sustainability. *Current Opinion in Environmental Sustainability*, *33*, 58–69. https://doi.org/10.1016/j.cosust.2018.04.009

Liu, J., Hull, V., Godfray, H. C. J., Tilman, D., Gleick, P., Hoff, H., Pahl-Wostl, C., Xu, Z., Chung, M. G., Sun, J., & Li, S. (2018). Nexus approaches to global sustainable development. *Nature Sustainability*, *1*(9), 466–476. https://doi.org/10.1038/s41893-018-0135-8

Liu, S., & Cook, D. (2016). Eradicate, contain, or live with it? Collaborating with stakeholders to evaluate responses to invasive species. *Food Security*, *8*(1), 49–59. https://doi.org/10.1007/s12571-015-0525-y

Liu, S., Proctor, W., & Cook, D. (2010). Using an integrated fuzzy set and deliberative multi-criteria evaluation approach to facilitate decision-making in invasive species management. *Ecological Economics*, *69*(12), 2374–2382. https://doi.org/10.1016/j.ecolecon.2010.07.004

Liu, S., Sheppard, A., Kriticos, D. J., & Cook, D. (2011). Incorporating uncertainty and social values in managing invasive alien species: A deliberative multi-criteria evaluation approach. *Biological Invasions*, *13*(10), 2323–2337. https://doi.org/10.1007/s10530-011-0045-4

Lodge, D. M., Williams, S., MacIsaac, H. J., Hayes, K. R., Leung, B., Reichard, S., Mack, R. N., Moyle, P. B., Smith, M., Andow, D. A., Carlton, J. T., & McMichael, A. (2006). Biological Invasions: Recommendations for U.S. Policy and Management. *Ecological Applications*, *16*(6), 2035–2054. JSTOR. https://doi.org/10.1890/1051-0761(2006)016[2035:BIRFUP]2.0.CO;2

Loorbach, D., Frantzeskaki, N., & Avelino, F. (2017). Sustainability transitions research: Transforming science and practice for societal change. *Annual Review of Environment and Resources*, *42*(1), 599–626. https://doi.org/10.1146/annurev-environ-102014-021340

López-Gómez, M. J., Aguilar-Perera, A., & Perera-Chan, L. (2014). Mayan diver-fishers as citizen scientists: Detection and monitoring of the invasive red lionfish in the Parque Nacional Arrecife Alacranes, southern Gulf of Mexico. *Biological Invasions*, *16*(7), 1351–1357. https://doi.org/10.1007/s10530-013-0582-0

Lopian, R. (2005). The International Plant Protection Convention and invasive alien species. In IPPC Secretariat (Ed.), *Identification of risks and management of invasive alien species using the IPPC framework* (p. 14). Food and Agriculture Organization of the United Nations. http://www.fao.org/3/y5968e/y5968e05.htm#bm05.5

Lowry, E., Rollinson, E. J., Laybourn, A. J., Scott, T. E., Aiello-Lammens, M. E., Gray, S. M., Mickley, J., & Gurevitch, J. (2013). Biological invasions: A field synopsis, systematic review, and database of the literature. *Ecology and Evolution*, *3*(1), 182–196. https://doi.org/10.1002/ece3.431

Lubeck, A. A., Metcalf, A. L., Beckman, C. L., Yung, L., & Angle, J. W. (2019). Collective factors drive individual invasive species control behaviors: Evidence from private lands in Montana, USA. *Ecology and Society*, *24*(2), 32. https://doi.org/10.5751/es-10897-240232

Lubell, M. (2004). Collaborative environmental institutions: All talk and no action? *Journal of Policy Analysis and Management*, *23*(3), 549–573. https://doi.org/10.1002/pam.20026

Lubell, M. (2013). Governing institutional complexity: The ecology of games framework. *Policy Studies Journal*, *41*(3), 537–559. https://doi.org/10.1111/psj.12028

Lubell, M., Henry, A. D., & McCoy, M. (2010). Collaborative Institutions in an Ecology of Games. *American Journal of Political Science*, *54*(2), 287–300. https://doi.org/10.1111/j.1540-5907.2010.00431.x

Lubell, M., Jasny, L., & Hastings, A. (2017). Network Governance for Invasive Species Management. *Conservation Letters*, *10*(6), 699–707. https://doi.org/10.1111/conl.12311

Lucy, F. E., Roy, H., Simpson, A., Carlton, J. T., Hanson, J. M., Magellan, K., Campbell, M. L., Costello, M. J., Pagad, S., Hewitt, C. L., McDonald, J., Cassey, P., Thomaz, S. M., Katsanevakis, S., Zenetos, A., Tricarico, E., Boggero, A., Groom, Q. J., Adriaens, T., … Panov, V. E. (2016). INVASIVESNET towards an international association for open knowledge on invasive alien species. *Management of Biological Invasions*, *7*(2), 131–139. https://doi.org/10.3391/mbi.2016.7.2.01

LXIV Ordinary Meeting. (2017). *SIM – Document Details*. https://documentos.mercosur.int/public/reuniones/doc/6499

Lynch, S. C. (2020). *Responding to an emergent plant pest-pathogen complex across social-ecological scales* [Doctoral Thesis, University of California, Santa Cruz]. https://escholarship.org/uc/item/19f5d5vk

Ma, Z., Clarke, M., & Church, S. P. (2018). Insights into individual and cooperative invasive plant management on family forestlands. *Land Use Policy*, *75*, 682–693. https://doi.org/10.1016/j.landusepol.2018.02.010

Macdonald, E. A., Hinks, A., Weiss, D. J., Dickman, A., Burnham, D., Sandom, C. J., Malhi, Y., & Macdonald, D. W. (2017). Identifying ambassador species for conservation marketing. *Global Ecology and Conservation*, *12*, 204–214. https://doi.org/10.1016/j.gecco.2017.11.006

Maclean, K., Farbotko, C., Mankad, A., Robinson, C. J., Curnock, M., Collins, K., & McAllister, R. R. (2018). Building social resilience to biological invasions. A case study of Panama Tropical Race 4 in the Australian Banana Industry. *Geoforum*, *97*, 95–105. https://doi.org/10.1016/j.geoforum.2018.10.018

Maclean, K., Robinson, C., Bock, E., & Rist, P. (2022). Reconciling risk and responsibility on Indigenous country: Bridging the boundaries to guide knowledge sharing for cross-cultural biosecurity risk management in northern Australia. *Journal of Cultural Geography*, *39*(1), 32–54. https://doi.org/10.1080/08873631.2021.1911078

Maggi, E., Benedetti-Cecchi, L., Castelli, A., Chatzinikolaou, E., Crowe, T. P., Ghedini, G., Kotta, J., Lyons, D. A., Ravaglioli, C., Rilov, G., Rindi, L., & Bulleri, F. (2015). Ecological impacts of invading seaweeds: A meta-analysis of their effects at different trophic levels. *Diversity and Distributions*, *21*(1), 1–12. https://doi.org/10.1111/ddi.12264

Mainka, S. A., & Howard, G. W. (2010). Climate change and invasive species: Double jeopardy. *Integrative Zoology*, *5*(2), 102–111. https://doi.org/10.1111/j.1749-4877.2010.00193.x

Manier, J., & Martin, A. (1998). *Park worker found beetle ID on web – Chicago Tribune*. https://www.chicagotribune.com/news/ct-xpm-1998-07-17-9807170143-story.html

Mannino, A. M., & Balistreri, P. (2018). Citizen science: A successful tool for monitoring invasive alien species (IAS) in Marine Protected Areas. The case study of the Egadi Islands MPA (Tyrrhenian Sea, Italy). *Biodiversity*, *19*(1–2), 42–48. https://doi.org/10.1080/14888386.2018.1468280

Mansouri, A., Cregut, M., Abbes, C., Durand, M.-J., Landoulsi, A., & Thouand, G. (2017). The Environmental Issues of DDT Pollution and Bioremediation: A Multidisciplinary Review. *Applied Biochemistry and Biotechnology*, *181*(1), 309–339. https://doi.org/10.1007/s12010-016-2214-5

Marchante, H., Morais, M. C., Gamela, A., & Marchante, E. (2017). Using a WebMapping Platform to Engage Volunteers to Collect Data on Invasive Plants Distribution. *Transactions in GIS*, *21*(2), 238–252. https://doi.org/10.1111/tgis.12198

Marire, J. (2015). The Political Economy of South African Trout Fisheries. *Journal of Economic Issues*, *49*(1), 47–70. https://doi.org/10.1080/00213624.2015.1013878

Marshall, G. R., Coleman, M. J., Sindel, B. M., Reeve, I. J., & Berney, P. J. (2016). Collective action in invasive species control, and prospects for community-based governance: The case of serrated tussock (*Nassella trichotoma*) in New South Wales, Australia. *Land Use Policy*, *56*, 100–111. https://doi.org/10.1016/j.landusepol.2016.04.028

Martin, D. M., Mazzotta, M., & Bousquin, J. (2018). Combining ecosystem services assessment with structured decision making to support ecological restoration planning. *Environmental Management*, *62*(3), 608–618. https://doi.org/10.1007/s00267-018-1038-1

Martin, L. J., Blossey, B., & Ellis, E. (2012). Mapping where ecologists work: Biases in the global distribution of terrestrial ecological observations. *Frontiers in Ecology and the Environment*, *10*(4), 195–201. https://doi.org/10.1890/110154

Martin, P., Le Gal, E., & Choy, D. L. (2016). Effect of Social and Institutional Fragmentation on Collective Action in Peri-Urban Settings. In B. Maheshwari, B. Thoradeniya, & V. P. Singh (Eds.), *Balanced Urban Development: Options and Strategies for Liveable Cities* (Vol. 72, pp. 465–480). Springer. https://doi.org/10.1007/978-3-319-28112-4_28

Martin, P., & Taylor, N. (2018). Environmental Stewardship Duties in Biosecurity: Issues and Challenges. *Environmental and Planning Law Journal*, *35*(6), 743–762. https://doi.org/10.3316/agispt.20181228005297

Martinez, B., Reaser, J. K., Dehgan, A., Zamft, B., Baisch, D., McCormick, C., Giordano, A. J., Aicher, R., & Selbe, S. (2020). Technology innovation: Advancing capacities for the early detection of and rapid response to invasive species. *Biological Invasions*, *22*(1), 75–100. https://doi.org/10.1007/s10530-019-02146-y

Mason, P. G., Cock, M. J. W., Barratt, B. I. P., Klapwijk, J. N., van Lenteren, J. C., Brodeur, J., Hoelmer, K. A., & Heimpel, G. E. (2018). Best practices for the use and exchange of invertebrate biological control genetic resources relevant for food and agriculture. *BioControl*, *63*(1), 149–154. https://doi.org/10.1007/s10526-017-9810-3

Mato-Amboage, R., Pitchford, J. W., & Touza, J. (2019). Public–private partnerships for biosecurity: An opportunity for risk sharing. *Journal of Agricultural Economics*, *70*(3), 771–788. https://doi.org/10.1111/1477-9552.12315

Mazza, G., Aquiloni, L., Inghilesi, A. F., Giuliani, C., Lazzaro, L., Ferretti, G., Lastrucci, L., Foggi, B., & Tricarico, E. (2015). Aliens just a click away: The online aquarium trade in Italy. *Management of Biological Invasions*, *6*(3), 253–261. https://doi.org/10.3391/mbi.2015.6.3.04

McAllister, R. R. J., Kruger, H., Stenekes, N., & Garrard, R. (2020). Multilevel stakeholder networks for Australian marine biosecurity: Well-structured for top-down information provision, requires better two-way communication. *Ecology and Society*, *25*(3). https://doi.org/10.5751/ES-11583-250318

McAllister, R. R. J., Robinson, C. J., Maclean, K., Guerrero, A. M., Collins, K., Taylor, B. M., & De Barro, P. J. (2015). From local to central: A network analysis of who manages plant pest and disease outbreaks across scales. *Ecology and Society*, *20*(1), 67. https://doi.org/10.5751/ES-07469-200167

McAusland, C., & Costello, C. (2004). Avoiding invasives: Trade-related policies for controlling unintentional exotic species introductions. *Journal of Environmental Economics and Management*, *48*(2), 954–977. https://doi.org/10.1016/j.jeem.2003.11.002

McBeath, J. H., & McBeath, J. (2010). Invasive Species and Food Security. In *Environmental Change and Food Security in China* (Vol. 35, pp. 157–176). Springer. https://doi.org/10.1007/978-1-4020-9180-3_6

McCarthy, A. H., Peck, L. S., Hughes, K. A., & Aldridge, D. C. (2019). Antarctica: The final frontier for marine biological invasions. *Global Change Biology*, *25*(7), 2221–2241. https://doi.org/10.1111/gcb.14600

McClelland, G. T. W., Altwegg, R., van Aarde, R. J., Ferreira, S., Burger, A. E., & Chown, S. L. (2018). Climate change leads to increasing population density and impacts of a key island invader. *Ecological Applications*, *28*(1), 212–224. https://doi.org/10.1002/eap.1642

McDermott, S. M. (2015). Optimal regulation of invasive species long-range spread: A general equilibrium approach. *The BE Journal of Economic Analysis & Policy*, *15*(4), 1731–1752. https://doi.org/10.1515/bejeap-2014-0087

McDermott, S. M., Irwin, R. E., & Taylor, B. W. (2013). Using economic instruments to develop effective management of invasive species: Insights from a bioeconomic model. *Ecological Applications*, *23*(5), 1086–1100. https://doi.org/10.1890/12-0649.1

McGeoch, M. A., Butchart, S. H. M., Spear, D., Marais, E., Kleynhans, E. J., Symes, A., Chanson, J., & Hoffmann, M. (2010). Global indicators of biological invasion: Species numbers, biodiversity impact and policy responses. *Diversity and Distributions*, *16*(1), 95–108. https://doi.org/10.1111/j.1472-4642.2009.00633.x

McGeoch, M. A., Genovesi, P., Bellingham, P. J., Costello, M. J., McGrannachan, C., & Sheppard, A. (2016). Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion. *Biological Invasions*, *18*(2), 299–314. https://doi.org/10.1007/s10530-015-1013-1

McGeoch, M. A., & Jetz, W. (2019). Measure and Reduce the Harm Caused by Biological Invasions. *One Earth*, *1*(2), 171–174. https://doi.org/10.1016/j.oneear.2019.10.003

McGeoch, M. A., Shaw, J. D., Terauds, A., Lee, J. E., & Chown, S. L. (2015). Monitoring biological invasion across the broader Antarctic: A baseline and indicator framework. *Global Environmental Change*, *32*, 108–125. https://doi.org/10.1016/j.gloenvcha.2014.12.012

McGeoch, M. A., Spear, D., Kleynhans, E. J., & Marais, E. (2012). Uncertainty in invasive alien species listing. *Ecological Applications*, *22*(3), 959–971. https://doi.org/10.1890/11-1252.1

McGeoch, M. A., & Squires, Z. E. (2015). *An essential variable approach to monitoring biological invasions: Guide for countries*. GEO BON Technical Series 2. https://www.geobon.org/downloads/biodiversity-monitoring/technical-reports/GEOBON/2015/Monitoring-Biological-Invasions.pdf

McGrannachan, C. M., Pagad, S., & McGeoch, M. A. (2021). A multiregional assessment of transnational pathways of introduction. *NeoBiota*, *64*, 43–67. https://doi.org/10.3897/neobiota.64.60642

McKiernan, S. (2018). Managing invasive plants in a rural-amenity landscape: The role of social capital and Landcare. *Journal of Environmental Planning and Management*, *61*(8), 1419–1437. https://doi.org/10.1080/09640568.2017.1351930

McKinley, D. C., Miller-Rushing, A. J., Ballard, H. L., Bonney, R., Brown, H., Cook-Patton, S. C., Evans, D. M., French, R. A., Parrish, J. K., Phillips, T. B., Ryan, S.

F., Shanley, L. A., Shirk, J. L., Stepenuck, K. F., Weltzin, J. F., Wiggins, A., Boyle, O. D., Briggs, R. D., Chapin, S. F., … Soukup, M. A. (2017). Citizen science can improve conservation science, natural resource management, and environmental protection. *Biological Conservation*, *208*, 15–28. https://doi.org/10.1016/j.biocon.2016.05.015

McLeod, L. J., & Saunders, G. R. (2011). Can legislation improve the effectiveness of fox control in NSW? *Australasian Journal of Environmental Management*, *18*(4), 248–259. https://doi.org/10.1080/14486563.2011.621409

Mcowen, C. J., Ivory, S., Dixon, M. J. R., Regan, E. C., Obrecht, A., Tittensor, D. P., Teller, A., & Chenery, A. M. (2016). Sufficiency and Suitability of Global Biodiversity Indicators for Monitoring Progress to 2020 Targets. *Conservation Letters*, *9*(6), 489–494. https://doi.org/10.1111/conl.12329

McQuilton, P., Gonzalez-Beltran, A., Rocca-Serra, P., Thurston, M., Lister, A., Maguire, E., & Sansone, S.-A. (2016). BioSharing: Curated and crowd-sourced metadata standards, databases and data policies in the life sciences. *Database*, *2016*, baw075. https://doi.org/10.1093/database/baw075

Meadows, D. H. (1999). *Leverage points: Places to intervene in a system*. The sustainability Institute. https://donellameadows.org/wp-content/userfiles/Leverage_Points.pdf

MERCOSUR. (2006). *Anexo III: Declaración De Los Ministros De Medio Ambiente Sobre Estrategia De Biodiversidad Del Mercosur*. https://ambiente.mercosur.int/MD_upload/Archivos/1/File/biblioteca/pdf/Biodiversidad/Estrategia_Biodiversidad_Mercosur.pdf

MERCOSUR. (2019). *Guidelines For The Development Of A Plan For The Prevention, Monitoring, Control And Mitigation Of Invasive Alien Specie*. https://normas.mercosur.int/public/normativas/3769

Messetchkova, I. (2021, November 2). *MDB Joint Nature Statement*. UN Climate Change Conference (COP26) at the SEC – Glasgow 2021. https://ukcop26.org/mdb-joint-statement/

Meyerson, L. A., & Simberloff, D. (Eds.). (2020). Special Issue: Early Detection and Rapid Response. *Biological Invasions*, *22*(1). https://link.springer.com/journal/10530/volumes-and-issues/22-1?fbclid=IwAR3cO07H9rnlnalHbCVrxDlu1tvRiWLAFT_BLMjPbhq9mIHzfe2RomI5Rww

Mineur, F., Cook, E. J., Minchin, D., Bohn, K., MacLeod, A., & Maggs, C. A. (2012). Changing coasts: Marine aliens and artificial structures. In R. N. Gibson, R. J. A. Atkinson, J. D. M. Gordon, & R. N. Hughes (Eds.), *Oceanography and Marine Biology: An Annual Review* (First edition, Vol. 50, pp. 189–233). CRC Press. https://www.taylorfrancis.com/chapters/edit/10.1201/b12157-6/changing-coasts-marine-aliens-arti%C2%8Dcial-structures-fr%C3%A9d%C3%A9ric-mineur-elizabeth-cook-dan-minchin-katrin-bohn-adrian-macleod-christine--maggs?context=ubx&refId=2930e715-4984-4ce8-81b6-66fa5cbd65e9

Ministério do Meio Ambiente/Secretaria de Biodiversidade. (2018). *Comissão Nacional De Biodiversidade Resolução Nº 7, De 29 De Maio De 2018* (p. 69). https://www.gov.br/mma/pt-br/assuntos/biodiversidade/fauna-e-flora/especies-exoticas-invasoras/resconabio072018estratgianacionalparaespciesexticasinvasoras.pdf

Miranda, R. J., Nunes, J. A. C. C., Creed, J. C., Barros, F., Macieira, R. M., Santos, R. G., Lima, G. V., Pontes, A. V. F., Silva, L. G. F. C., Cordeiro, R. T., Sampaio, C. L. S., Pinto, T. K., Malhado, A. C. M., Ladle, R., & Pereira, P. H. (2020). Brazil policy invites marine invasive species. *Science*, *368*(6490), 481. https://doi.org/10.1126/science.abb7255

Mitchell, P. D., Brown, Z., & McRoberts, N. (2018). Economic issues to consider for gene drives. *Journal of Responsible Innovation*, *5*(sup1), S180–S202. https://doi.org/10.1080/23299460.2017.1407914

Miyanaga, K., & Nakai, K. (2021). Making adaptive governance work in biodiversity conservation: Lessons in invasive alien aquatic plant management in Lake Biwa, Japan. *Ecology and Society*, *26*(2), 11. https://doi.org/10.5751/ES-12352-260211

Molloy, J. C. (2011). The Open Knowledge Foundation: Open Data Means Better Science. *PLoS Biology*, *9*(12), e1001195. https://doi.org/10.1371/journal.pbio.1001195

Monaco, A. & IUCN Invasive Species Specialist Group. (2021, August). *Guidance Document on E-Commerce and IAS*. 41st meeting of Convention on The Conservation of European Wildlife and Natural Habitats, Strasbourg. https://rm.coe.int/inf39e-rev-2021-guidance-document-e-commerce-and-ias/1680a4b7f3

Moon, K., Blackman, D. A., & Brewer, T. D. (2015). Understanding and integrating knowledge to improve invasive species management. *Biological Invasions*, *17*(9), 2675–2689. https://doi.org/10.1007/s10530-015-0904-5

Moon, K., Blackman, D., Brewer, T. D., & Sarre, S. D. (2017). Environmental governance for urgent and uncertain problems. *Biological Invasions*, *19*(3), 785–797. https://doi.org/10.1007/s10530-016-1351-7

Mooney, H. A., & Hobbs, R. J. (2000). Global change and invasive species: Where do we go from here. In H. A. Moone & R. J. Hobbs (Eds.), *Invasive species in a changing world* (pp. 425–434). Island Press.

Morrisey, D., Inglis, G., Neil, K., Bradley, A., & Fitridge, I. (2011). Characterization of the marine aquarium trade and management of associated marine pests in Australia, a country with stringent import biosecurity regulation. *Environmental Conservation*, *38*(1), 89–100. https://doi.org/10.1017/S0376892911000014

Muller, A., Schader, C., El-Hage Scialabba, N., Brüggemann, J., Isensee, A., Erb, K.-H., Smith, P., Klocke, P., Leiber, F., Stolze, M., & Niggli, U. (2017). Strategies for feeding the world more sustainably with organic agriculture. *Nature Communications*, *8*(1), 1290. https://doi.org/10.1038/s41467-017-01410-w

Muller, S., Power, E. R., Suchet-Pearson, S., Wright, S., & Lloyd, K. (2009). "Quarantine Matters!": Quotidian Relationships around Quarantine in Australia's Northern Borderlands. *Environment and Planning A: Economy and Space*, *41*(4), 780–795. https://doi.org/10.1068/a40196

Mulligan, S., & Stoett, P. (2000). A Global Bio-Prospecting Regime: Partnership or Piracy? *International Journal*, *55*(2), 224–246. https://doi.org/10.2307/40203478

Mumba, M., & Thompson, J. R. (2005). Hydrological and ecological impacts of dams on the Kafue Flats floodplain system, southern Zambia. *Physics and Chemistry of the Earth, Parts A/B/C*, *30*(6–7), 442–447. https://doi.org/10.1016/j.pce.2005.06.009

Munakamwe, Z., Constantine, A., & McInerney, C. (2018). The Victorian weed spotters: Recruitment and training of citizen scientists. *Proceedings of the 21st Australasian Weeds Conference*, 9–13. https://caws.org.nz/old-site/awc/2018/awc201811931.pdf

Mungi, N. A., Qureshi, Q., & Jhala, Y. V. (2020). Expanding niche and degrading forests: Key to the successful global invasion of *Lantana camara* (*sensu* lato). *Global Ecology and Conservation*, *23*, e01080. https://doi.org/10.1016/j.gecco.2020.e01080

Mungi, N. A., Qureshi, Q., & Jhala, Y. V. (2021). Role of species richness and human impacts in resisting invasive species in tropical forests. *Journal of Ecology*, *109*(9), 3308–3321. https://doi.org/10.1111/1365-2745.13751

Muñoz, E. N. (2016). *Caulerpa Conquest: A Biological Eradication on the California Coast*. Open Books Press.

Murphy, S. T., & Cheesman, O. D. (2006). *The Aid Trade—International Assistance Programs as Pathways for the Introduction of Invasive Alien Species* (Biodiversity Series, p. 50) [A Preliminary Report]. World Bank Environmental Department.

MVOTMA, Ministerio de Relaciones Exteriores, & UNDP. (2016). *Estrategia Nacional para la Conservación y Uso Sostenible de la Diversidad Biológica del Uruguay 2016-2020*. Ministerio de Vivienda, Ordenamiento Territorial y Medio Ambiente.

NAPRA Network. (2010). Risk assessment scheme for non-native species in Great Britain. *Non-Native Species Secretariat*.

Natural Justice. (2022). *Biocultural Community Protocols*. Natural Justice. https://naturaljustice.org/publication/biocultural-community-protocols/

Neale, T., & Macdonald, J. M. (2019). Permits to burn: Weeds, slow violence, and the extractive future of northern Australia. *Australian Geographer*, *50*(4), 417–433. https://doi.org/10.1080/00049182.2019.1686731

Nel, J. L., Roux, D. J., Driver, A., Hill, L., Maherry, A. C., Snaddon, K., Petersen, C. R., Smith-Adao, L. B., Van Deventer, H., & Reyers, B. (2016). Knowledge co-production and boundary work to promote implementation of conservation plans. *Conservation Biology*, *30*(1), 176–188. https://doi.org/10.1111/cobi.12560

Newig, J., Challies, E., Jager, N. W., Kochskaemper, E., & Adzersen, A. (2018). The Environmental Performance of Participatory and Collaborative Governance: A Framework of Causal Mechanisms. *Policy Studies Journal*, *46*(2), 269–297. https://doi.org/10.1111/psj.12209

Nguyen, V. H. (2007). Vietnam's catfish and marine shrimp production: An example of growth and sustainability issues. *AQUA Culture Asia Pacific Magazine*, *3*(4), 36–38. https://aquaasiapac.com/issue/july-aug-2007/

Niemiec, R. M., Ardoin, N. M., Wharton, C. B., & Asner, G. P. (2016). Motivating residents to combat invasive species on private lands: Social norms and community reciprocity. *Ecology and Society*, *21*(2), 30. https://doi.org/10.5751/ES-08362-210230

Niemiec, R. M., Willer, R., Ardoin, N. M., & Brewer, F. K. (2019). Motivating landowners to recruit neighbors for private land conservation. *Conservation Biology: The Journal of the Society for Conservation Biology*, *33*(4), 930–941. https://doi.org/10.1111/cobi.13294

NISC. (2008). *2008-2012 National Invasive Species Management Plan*. National Invasive Species Council. https://www.landcan.org/pdfs/2008-2012%20National%20Invasive%20Species%20Management%20Plan.pdf

NISC. (2016). *Management Plan: 2016–2018*. National Invasive Species Council. https://www.doi.gov/sites/doi.gov/files/uploads/2016-2018-nisc-management-plan.pdf

Niyazov, Y., Vogel, C., Price, R., Lund, B., Judd, D., Akil, A., Mortonson, M., Schwartzman, J., & Shron, M. (2016). Open Access Meets Discoverability: Citations to Articles Posted to Academia.edu. *PLoS ONE*, *11*(2), e0148257. https://doi.org/10.1371/journal.pone.0148257

Nourani, S. W., Decker, D. J., & Krasny, M. E. (2019). Extension as a Multilevel Bridging Organization: Supporting Networked Environmental Governance. *Journal of Extension*, *57*(5). https://archives.joe.org/joe/2019october/a2.php

Nourani, S. W., Krasny, M. E., & Decker, D. J. (2018). Learning and linking for invasive species management. *Ecology and Society*, *23*(3), 29. https://doi.org/10.5751/ES-10327-230329

Novoa, A., Kaplan, H., Wilson, J. R. U., & Richardson, D. M. (2016). Resolving a Prickly Situation: Involving Stakeholders in Invasive Cactus Management in South Africa. *Environmental Management*, *57*(5), 998–1008. https://doi.org/10.1007/s00267-015-0645-3

Nuñez, M. A., & Amano, T. (2021). Monolingual searches can limit and bias results in global literature reviews. *Nature Ecology & Evolution*, *5*(3), 264. https://doi.org/10.1038/s41559-020-01369-w

Nuñez, M. A., Chiuffo, M. C., Seebens, H., Kuebbing, S., McCary, M. A., Lieurance, D., Zhang, B., Simberloff, D., & Meyerson, L. A. (2021). Two decades of data reveal that Biological Invasions needs to increase participation beyond North America, Europe, and Australasia. *Biological Invasions*, *24*, 333–340. https://doi.org/10.1007/s10530-021-02666-6

OECD. (2003). *OECD Environmental Indicators. Development, measurement and use* (OECD Environmental Indicators) [Reference paper]. https://www.oecd.org/env/indicators-modelling-outlooks/24993546.pdf

OECD. (2007). *Subsidy Reform and Sustainable Development: Political Economy Aspects*. Organisation for Economic Co-operation and Development. https://www.cbd.int/financial/fiscalenviron/g-subsidyreform-oecd.pdf

OECD. (2017). *Towards a G7 target to phase out environmentally harmful subsidies*. https://www.minambiente.it/sites/default/files/archivio/allegati/sviluppo_sostenibile/background_paper_4_G7_env_OECD_Towards_G7_target_to_phase_out_EHSs.pdf

OECD. (2019). *The Post-2020 Biodiversity Framework: Targets, indicators and measurability implications at global and national level* [November version]. https://www.oecd.org/environment/resources/biodiversity/report-the-post-2020-biodiversity-framework-targets-indicators-and-measurability-implications-at-global-and-national-level.pdf

OECD. (2020). *Trade and the environment—OECD*. https://www.oecd.org/trade/topics/trade-and-the-environment/

Ojaveer, H., Galil, B. S., Campbell, M. L., Carlton, J. T., Canning-Clode, J., Cook, E. J., Davidson, A. D., Hewitt, C. L., Jelmert, A., Marchini, A., McKenzie, C. H., Minchin, D., Occhipinti-Ambrogi, A., Olenin, S., & Ruiz, G. (2015). Classification of non-indigenous species based on their impacts: Considerations for application in marine management. *PLoS Biology*, *13*(4), e1002130. https://doi.org/10.1371/journal.pbio.1002130

Ojaveer, H., Galil, B. S., Carlton, J. T., Alleway, H., Goulletquer, P., Lehtiniemi, M., Marchini, A., Miller, W., Occhipinti-Ambrogi, A., Peharda, M., Ruiz, G. M., Williams, S. L., & Zaiko, A. (2018). Historical baselines in marine bioinvasions: Implications for policy and management. *PLoS ONE*, *13*(8), e0202383. https://doi.org/10.1371/journal.pone.0202383

Olden, J. D., Poff, N. L., Douglas, M. R., Douglas, M. E., & Fausch, K. D. (2004). Ecological and evolutionary consequences of biotic homogenization. *Trends in Ecology & Evolution*, *19*(1), Article 1. https://doi.org/10.1016/j.tree.2003.09.010

Oldoni, D., Groom, Q., Adriaens, T., Davis, A. J. S., Reyserhove, L., Strubbe, D., Vanderhoeven, S., & Desmet, P. (2020). *Occurrence cubes: A new paradigm for aggregating species occurrence data* [Preprint]. Ecology. https://doi.org/10.1101/2020.03.23.983601

Olenin, S., Narščius, A., Minchin, D., David, M., Galil, B. S., Gollasch, S., Marchini, A., Occhipinti-Ambrogi, A., Ojaveer, H., & Zaiko, A. (2014). Making non-indigenous species information systems practical for management and useful for research: An aquatic perspective. *Biological Conservation*, *173*, 98–107. https://doi.org/10.1016/j.biocon.2013.07.040

Omondiagbe, H. A., Towns, D. R., Wood, J. K., & Bollard-Breen, B. (2017). Stakeholders and social networks identify potential roles of communities in sustainable management of invasive species. *Biological Invasions*, *19*(10), 3037–3049. https://doi.org/10.1007/s10530-017-1506-1

Open Knowledge Foundation. (2021). *Open Definition—Defining Open in Open Data, Open Content and Open Knowledge*. https://opendefinition.org/

Ostrom, E. (2010). Polycentric systems for coping with collective action and global environmental change. *Global Environmental Change*, *20*(4), 550–557. https://doi.org/10.1016/j.gloenvcha.2010.07.004

Oswalt, S., Oswalt, C., Crall, A., Rabaglia, R., Schwartz, M. K., Kerns, B. K., Poland, T. M., Patel-Weynand, T., Finch, D. M., Ford Miniat, C., Hayes, D. C., & Lopez, V. M. (2021). Inventory and Monitoring of Invasive Species. In *Invasive Species in Forests and Rangelands of the United States: A Comprehensive Science Synthesis for the United States Forest Sector* (pp. 231–242). Springer International Publishing. https://doi.org/10.1007/978-3-030-45367-1_10

Outhwaite, O. (2010). The International Legal Framework For Biosecurity And The Challenges Ahead. *Review of European Community & International Environmental Law*, *19*(2), 207–226. https://doi.org/10.1111/j.1467-9388.2010.00678.x

Outhwaite, O. (2013). Legal frameworks for biosecurity. In A. Dobson, K. Barker, & S. L. Taylor (Eds.), *Biosecurity: The Socio-Politics of Invasive Species and Infectious Diseases* (pp. 89–104). Routledge.

Outhwaite, O. (2017). Neither Fish, nor Fowl: Honeybees and the Parameters of Current Legal Frameworks for Animals, Wildlife and Biodiversity. *Journal of Environmental Law*, *29*(2), 317–341. https://doi.org/10.1093/jel/eqx005

Outhwaite, O. (2018). Biosecurity, invasive species and the law. In C. R. McManis & B. Ong (Eds.), *Routledge handbook of biodiversity and the law* (1st edition., pp. 83–97). Routledge. https://www.taylorfrancis.com/chapters/edit/10.4324/9781315530857-6/biosecurity-invasive-species-law-opi-outhwaite

Packer, J. G., Meyerson, L. A., Richardson, D. M., Brundu, G., Allen, W. J., Bhattarai, G. P., Brix, H., Canavan, S., Castiglione, S., Cicatelli, A., Čuda, J., Cronin, J. T., Eller, F., Guarino, F., Guo, W.-H., Guo, W.-Y., Guo, X., Hierro, J. L., Lambertini, C., … Pyšek, P. (2017). Global networks for invasion science: Benefits, challenges and guidelines. *Biological Invasions*, *19*(4), 1081–1096. https://doi.org/10.1007/s10530-016-1302-3

Padilla, D. K., & Williams, S. L. (2004). Beyond ballast water: Aquarium and ornamental trades as sources of invasive species in aquatic ecosystems. *Frontiers in Ecology and the Environment*, *2*(3), 131–138. https://doi.org/10.1890/1540-9295(2004)002[0131:BBWAAO]2.0.CO;2

Padt, F., & Arts, B. (2014). Concepts of scale. In F. Padt, P. Opdam, N. Polman, & C. Termeer (Eds.), *Scale-sensitive governance of the environment* (pp. 1–13). John Wiley & Sons. https://doi.org/10.1002/9781118567135.ch1

Padt, F., Opdam, P. F. M., Polman, N. B. P., & Termeer, C. J. A. M. (Eds.). (2014). *Scale-sensitive governance of the environment*. John Wiley & Sons. https://doi.org/10.1002/9781118567135

Pagad, S., Affleck, S., & McGeoch, M. A. (2020). *International Adoption of Invasive Alien Species Policy* (p. 3420679 Bytes). La Trobe. https://doi.org/10.26181/5F7EA2D538147

Pagad, S., Bisset, S., Genovesi, P., Groom, Q., Hirsch, T., Jetz, W., Ranipeta, A., Schigel, D., Sica, Y. V., & McGeoch, M. A. (2022). Country Compendium of the Global Register of Introduced and Invasive Species. *Scientific Data*, *9*(1), 391. https://doi.org/10.1038/s41597-022-01514-z

Pagad, S., Genovesi, P., Carnevali, L., Scalera, R., & Clout, M. (2015). IUCN SSC Invasive Species Specialist Group: Invasive alien species information management supporting practitioners, policy makers and decision takers. *Management of Biological Invasions*, *6*(2), 127–135. https://doi.org/10.3391/mbi.2015.6.2.03

Pagad, S., Genovesi, P., Carnevali, L., Schigel, D., & McGeoch, M. A. (2018). Introducing the Global Register of Introduced and Invasive Species. *Scientific Data*, *5*(1), 170202. https://doi.org/10.1038/sdata.2017.202

Paini, D. R., Sheppard, A. W., Cook, D. C., De Barro, P. J., Worner, S. P., & Thomas, M. B. (2016). Global threat to agriculture from invasive species. *Proceedings of the National Academy of Sciences of the United States of America*, *113*(27), 7575–7579. https://doi.org/10.1073/pnas.1602205113

Palacios, X. F. (1997). Contribution to the estimation of countries' interdependence in the area of plant genetic resources. *Comision de Recursos Geneticos Para La Alimentacion y La Agricultura. Reunion Extraordinaria. 4, Rome (Italy), 1-5 Dec 1997*.

Palmer, J. R. B., Oltra, A., Collantes, F., Delgado, J. A., Lucientes, J., Delacour, S., Bengoa, M., Eritja, R., & Bartumeus, F. (2017). Citizen science provides a reliable and scalable tool to track disease-carrying mosquitoes. *Nature Communications*, *8*(1), 916. https://doi.org/10.1038/s41467-017-00914-9

Paola, M. di, & Kravetz, D. (2004). Invasive alien species: Legal and institutional framework in Argentina. *Environmental Law Institute*, 71–87. https://www.ecolex.org/details/literature/invasive-alien-species-legal-and-institutional-framework-in-argentina-ana-072038/

Pasiecznik, N. (1999). Prosopis-pest or providence, weed or wonder tree? *European Tropical Forest Research Network Newsletter*, *28*, 12–14. https://agris.fao.org/agris-search/search.do?recordID=GB2012108487

Patterson, J. J., Thaler, T., Hoffmann, M., Hughes, S., Oels, A., Chu, E., Mert, A., Huitema, D., Burch, S., & Jordan, A. (2018). Political feasibility of 1.5 C societal transformations: The role of social justice. *Current Opinion in Environmental Sustainability*, *31*, 1–9. https://doi.org/10.1016/j.cosust.2017.11.002

Pauchard, A., Quiroz, C., García, R., Anderson, C. B., & Arroyo, M. T. K. (2011). Invasiones biológicas en América Latina y el Caribe: Tendencias en investigación para la conservación. In *Conservación Biológica: Perspectivas desde América Latina* (p. 17). Editorial Universitaria.

Pecl, G. T., Ogier, E., Jennings, S., van Putten, I., Crawford, C., Fogarty, H., Frusher, S., Hobday, A. J., Keane, J., Lee, E., MacLeod, C., Mundy, C., Stuart-Smith, J., & Tracey, S. (2019). Autonomous adaptation to climate-driven change in marine biodiversity in a global marine hotspot. *Ambio*, *48*(12), 1498–1515. https://doi.org/10.1007/s13280-019-01186-x

Peltzer, D. A., Bellingham, P. J., Dickie, I. A., Houliston, G., Hulme, P. E., Lyver, P. O., McGlone, M., Richardson, S. J., & Wood, J. (2019). Scale and complexity implications of making New Zealand predator-free by 2050. *Journal of the Royal Society of New Zealand*, *49*(3), 412–439. https://doi.org/10.1080/03036758.2019.1653940

Pereira, H. M., Ferrier, S., Walters, M., Geller, G. N., Jongman, R. H. G., Scholes, R. J., Bruford, M. W., Brummitt, N., Butchart, S. H. M., Cardoso, A. C., Coops, N. C., Dulloo, E., Faith, D. P., Freyhof, J., Gregory, R. D., Heip, C., Hoft, R., Hurtt, G., Jetz, W., … Wegmann, M. (2013). Essential Biodiversity Variables. *Science*, *339*(6117), 277–278. https://doi.org/10.1126/science.1229931

Pérez, J. E., Alfonsi, C., Nirchio, M., Muñoz, C., & Gómez, J. A. (2003). The introduction of exotic species in aquaculture: A solution or part of the problem? *Interciencia*, *28*(4), 234–238. https://www.researchgate.net/publication/233808881_The_introduction_of_exotic_species_in_aquaculture_A_solution_or_part_of_the_problem

Perrings, C., Burgiel, S., Lonsdale, M., Mooney, H., & Williamson, M. (2010). International cooperation in the solution to trade-related invasive species risks. *Annals of the New York Academy of Sciences*, *1195*(1), 198–212. https://doi.org/10.1111/j.1749-6632.2010.05453.x

Perrings, C., Williamson, M., Barbier, E. B., Delfino, D., Dalmazzone, S., Shogren, J., Simmons, P., & Watkinson, A. (2002). Biological Invasion Risks and the Public Good: An Economic Perspective. *Conservation Ecology*, *6*(1). https://www.jstor.org/stable/26271860

Perry, D., & Perry, G. (2008). Improving interactions between animal rights groups and conservation biologists. *Conservation Biology*, *22*(1), 27–35. https://doi.org/10.1111/j.1523-1739.2007.00845.x

Pertierra, L. R., Bartlett, J. C., Duffy, G. A., Vega, G. C., Hughes, K. A., Hayward, S. A. L., Convey, P., Olalla-Tarraga, M. A., & Aragón, P. (2020). Combining correlative and mechanistic niche models with human activity data to elucidate the invasive potential of a sub-Antarctic insect. *Journal of Biogeography*, *47*(3), 658–673. https://doi.org/10.1111/jbi.13780

Pheloung, P. C., Williams, P. A., & Halloy, S. R. (1999). A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. *Journal of Environmental Management*, *57*(4), 239–251. https://doi.org/10.1006/jema.1999.0297

Pidot, J. R. (2005). The Applicability of Nuisance Law to Invasive Plants: Can Common Law Liability Inspire Government Action? *Virginia Environmental Law Journal*, *24*(2), 183–230. https://www.jstor.org/stable/24787332

Pimentel, D., Zuniga, R., & Morrison, D. (2005). Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics*, *52*(3), 273–288. https://doi.org/10.1016/j.ecolecon.2004.10.002

Piria, M., Copp, G. H., Dick, J. T., Duplić, A., Groom, Q., Jelić, D., Lucy, F. E., Roy, H. E., Sarat, E., Simonović, P., Tomljanović, T., Tricarico, E., Weinlander, M., Adámek, Z., Bedolfe, S., Coughlan, N. E., Davis, E., Dobrzycka-Krahel, A., Grgić, Z., … Caffrey, J. M. (2017). Tackling invasive alien species in Europe II: threats and opportunities until 2020. *Management of Biological Invasions*, *8*(3), 273–286. https://doi.org/10.3391/mbi.2017.8.3.02

Piwowar, H., Priem, J., Larivière, V., Alperin, J. P., Matthias, L., Norlander, B., Farley, A., West, J., & Haustein, S. (2018). The state of OA: A large-scale analysis of the prevalence and impact of Open Access articles. *PeerJ*, *6*, e4375. https://doi.org/10.7717/peerj.4375

Plummer, R., Armitage, D. R., & de Loë, R. C. (2013). Adaptive Comanagement and Its Relationship to Environmental Governance. *Ecology and Society*, *18*(1), 21. https://doi.org/10.5751/ES-05383-180121

Plummer, R., Crona, B., Armitage, D. R., Olsson, P., Tengö, M., & Yudina, O. (2012). Adaptive Comanagement: A Systematic Review and Analysis. *Ecology and Society*, *17*(3), 11. https://doi.org/10.5751/ES-04952-170311

PNW-IPC. (2018). *Home*. Pacific Northwest Invasive Plant Council. https://www.pnw-ipc.org

Pochon, X., Bott, N. J., Smith, K. F., & Wood, S. A. (2013). Evaluating Detection Limits of Next-Generation Sequencing for the Surveillance and Monitoring of International Marine Pests. *PLoS ONE*, *8*(9), e73935. https://doi.org/10.1371/journal.pone.0073935

Pocock, M. J. O., Chandler, M., Bonney, R., Thornhill, I., Albin, A., August, T., Bachman, S., Brown, P. M. J., Cunha, D. G. F., Grez, A., Jackson, C., Peters, M., Rabarijaon, N. R., Roy, H. E., Zaviezo, T., & Danielsen, F. (2018). Chapter Six—A Vision for Global Biodiversity Monitoring With Citizen Science. In D. A. Bohan, A. J. Dumbrell, G. Woodward, & M. Jackson (Eds.), *Advances in Ecological Research* (Vol. 59, pp. 169–223). Academic Press. https://doi.org/10.1016/bs.aecr.2018.06.003

Pocock, M. J. O., Roy, H. E., August, T., Kuria, A., Barasa, F., Bett, J., Githiru, M., Kairo, J., Kimani, J., Kinuthia, W., Kissui, B., Madindou, I., Mbogo, K., Mirembe, J., Mugo, P., Muniale, F. M., Njoroge, P., Njuguna, E. G., Olendo, M. I., … Trevelyan, R. (2019). Developing the global potential of citizen science: Assessing opportunities that benefit people, society and the environment in East Africa. *Journal of Applied Ecology*, *56*(2), 274–281. https://doi.org/10.1111/1365-2664.13279

Poland, T. M., Emery, M. R., Ciaramitaro, T., Pigeon, E., & Pigeon, A. (2017). Emerald ash borer, black ash, and Native American basketmaking: Invasive insects, forest ecosystems, and cultural practices. In E. Freedman & M. Neuzil (Eds.), *Biodiversity, Conservation, and Environmental Management in the Great Lakes Basin* (pp. 127–140). Routledge. https://doi.org/10.4324/9781315268774

Poland, T. M., & Rassati, D. (2019). Improved biosecurity surveillance of non-native forest insects: A review of current methods. *Journal of Pest Science*, *92*(1), 37–49. https://doi.org/10.1007/s10340-018-1004-y

Praseeda Sanu, V., & Newport, J. K. (2010). Invasive alien species dispersal: The millennium biodiversity disaster. *Disaster Prevention and Management*, *19*(3), 291–297. https://doi.org/10.1108/09653561011052475

Primmer, E., Jokinen, P., Blicharska, M., Barton, D. N., Bugter, R., & Potschin, M. (2015). Governance of Ecosystem Services: A framework for empirical analysis.

Ecosystem Services, *16*, 158–166. https://doi.org/10.1016/j.ecoser.2015.05.002

pro-iBiosphere Consortium. (2014). *The Bouchout Declaration for Open Biodiversity Knowledge Management*. http://www.bouchoutdeclaration.org/

Provan, K. G., & Kenis, P. (2008). Modes of Network Governance: Structure, Management, and Effectiveness. *Journal of Public Administration Research and Theory*, *18*(2), 229–252. https://doi.org/10.1093/jopart/mum015

Pyšek, P., Hulme, P. E., Meyerson, L. A., Smith, G. F., Boatwright, J. S., Crouch, N. R., Figueiredo, E., Foxcroft, L. C., Jarošík, V., Richardson, D. M., Suda, J., & Wilson, J. R. U. (2013). Hitting the right target: Taxonomic challenges for, and of, plant invasions. *AoB PLANTS*, *5*, plt042. https://doi.org/10.1093/aobpla/plt042

Pyšek, P., Pergl, J., Dawson, W., Essl, F., Kreft, H., Weigelt, P., Winter, M., & van Kleunen, M. (2020). European plant invasions. In D. R. Clements, M. K. Upadhyaya, S. Joshi, & A. Shrestha (Eds.), *Global Plant Invasions* (pp. 151–165). Springer. https://doi.org/10.1007/978-3-030-89684-3_7

Pyšek, P., Richardson, D. M., Pergl, J., Jarošík, V., Sixtová, Z., & Weber, E. (2008). Geographical and taxonomic biases in invasion ecology. *Trends in Ecology & Evolution*, *23*(5), 237–244. https://doi.org/10.1016/j.tree.2008.02.002

Queloz, V., Grünig, C. R., Berndt, R., Kowalski, T., Sieber, T. N., & Holdenrieder, O. (2011). Cryptic speciation in *Hymenoscyphus albidus*. *Forest Pathology*, *41*(2), 133–142. https://doi.org/10.1111/j.1439-0329.2010.00645.x

Quirós, R. (1990). The Paraná river basin development and the changes in the lower basin fisheries. *Interciencia*, *15*(6), 442–451. https://scholar.google.com/scholar_lookup?title=The+Paran%C3%A1+river+basin+development+and+the+changes+in+the+lower+basin+fisheries&author=Quir%C3%B3s,+R.&publication_year=1990&journal=Interciencia&volume=15&pages=442%E2%80%93451

Ramírez-Albores, J. E., Badano, E. I., Flores, J., Flores-Flores, J. L., & Yáñez-Espinosa, L. (2019). Scientific literature on invasive alien species in a megadiverse country: Advances and challenges in Mexico. *NeoBiota*, *48*, 113–127. https://doi.org/10.3897/neobiota.48.36201

Ray, R. (2021). Who Controls Multilateral Development Finance? *Global Governance: A Review of Multilateralism and International Organizations*, *27*(1), 118–143. https://doi.org/10.1163/19426720-02701006

RDA. (2019). *International Indigenous Data Sovereignty IG*. Research Data Alliance. https://www.rd-alliance.org/groups/international-indigenous-data-sovereignty-ig

Redpath, S. M., Young, J., Evely, A., Adams, W. M., Sutherland, W. J., Whitehouse, A., Amar, A., Lambert, R. A., Linnell, J. D. C., Watt, A., & Gutiérrez, R. J. (2013). Understanding and managing conservation conflicts. *Trends in Ecology and Evolution*, *28*(2), 100–109. https://doi.org/10.1016/j.tree.2012.08.021

Reed, J., Van Vianen, J., Deakin, E. L., Barlow, J., & Sunderland, T. (2016). Integrated landscape approaches to managing social and environmental issues in the tropics: Learning from the past to guide the future. *Global Change Biology*, *22*(7), 2540–2554. https://doi.org/10.1111/gcb.13284

Reed, M. S. (2008). Stakeholder participation for environmental management: A literature review. *Biological Conservation*, *141*(10), 2417–2431. https://doi.org/10.1016/J.BIOCON.2008.07.014

Reed, M. S., Bonn, A., Slee, W., Beharry-Borg, N., Birch, J., Brown, I., Burt, T. P., Chapman, D., Chapman, P. J., Clay, G. D., Cornell, S. J., Fraser, E. D. G., Glass, J. H., Holden, J., Hodgson, J. A., Hubacek, K., Irvine, B., Jin, N., Kirkby, M. J., … Worrall, F. (2009). The Future of the Uplands. *Land Use Policy*, *26*(1), S204–S216. https://doi.org/10.1016/j.landusepol.2009.09.013

Reed, M. S., & Curzon, R. (2015). Stakeholder mapping for the governance of biosecurity: A literature review. *Journal of Integrative Environmental Sciences*, *12*(1), 15–38. https://doi.org/10.1080/1943815X.2014.975723

Reed, M. S., Vella, S., Challies, E., de Vente, J., Frewer, L., Hohenwallner-Ries, D., Huber, T., Neumann, R. K., Oughton, E. A., Sidoli del Ceno, J., & van Delden, H. (2018). A theory of participation: What makes stakeholder and public engagement in environmental management work? *Restoration Ecology*, *26*(S1), S7–S17. https://doi.org/10.1111/rec.12541

Reichard, S., Schmitz, C., Simberloff, D., Morrison, D., Lehtonen, P. P., Windle, P. N., Chavarría, G., & Mezitt, R. W. (2005). The tragedy of the commons revisited: Invasive species. *Frontiers in Ecology and the Environment*, *3*(2), 109–115. https://doi.org/10.1890/1540-9295(2005)003[0103:TTOTCR]2.0.CO;2

Reichman, O. J., Jones, M. B., & Schildhauer, M. P. (2011). Challenges and Opportunities of Open Data in Ecology. *Science*, *331*(6018), 703–705. https://doi.org/10.1126/science.1197962

Reid, M. J., Hull, L. A., Alter, T. R., Adams, L. B., Kleinert, H. M., & Woolnough, A. P. (2019). New development: Public sector responses to complex socio-ecological issues—no silver bullets for rabbits. *Public Money & Management*, *41*(4), 351–355. https://doi.org/10.1080/09540962.2019.1685168

Reo, N. J., Whyte, K., Ranco, D., Brandt, J., Blackmer, E., & Elliott, B. (2017). Invasive Species, Indigenous Stewards, and Vulnerability Discourse. *American Indian Quarterly*, *41*(3), 201–223. https://doi.org/10.5250/amerindiquar.41.3.0201

Reyserhove, L., Desmet, P., Oldoni, D., Adriaens, T., Strubbe, D., Davis, A. J. S., Vanderhoeven, S., Verloove, F., & Groom, Q. (2020). A checklist recipe: Making species data open and FAIR. *Database*, *2020*, baaa084. https://doi.org/10.1093/database/baaa084

Ricciardi, A., Blackburn, T. M., Carlton, J. T., Dick, J. T. A., Hulme, P. E., Iacarella, J. C., Jeschke, J. M., Liebhold, A. M., Lockwood, J. L., MacIsaac, H. J., Pyšek, P., Richardson, D. M., Ruiz, G. M., Simberloff, D., Sutherland, W. J., Wardle, D. A., & Aldridge, D. C. (2017). Invasion Science: A Horizon Scan of Emerging Challenges and Opportunities. *Trends in Ecology & Evolution*, *32*(6), 464–474. https://doi.org/10.1016/j.tree.2017.03.007

Ricciardi, A., Steiner, W. W. M., Mack, R. N., & Simberloff, D. (2000). Toward a Global Information System for Invasive Species. *BioScience*, *50*(3), 239–244. https://doi.org/10.1641/0006-3568(2000)050[0239:TAGISF]2.3.CO;2

Richards, T. J., Ellsworth, P., Tronstad, R., & Naranjo, S. (2010). Market-Based Instruments for the Optimal Control of Invasive Insect Species: *B. Tabaci* in Arizona. *Journal of Agricultural and Resource Economics*, *35*(3), 349–367. https://doi.org/10.22004/ag.econ.97852

Richardson, D. M. (1998). Forestry Trees as Invasive Aliens. *Conservation Biology*, *12*(1), 18–26. https://doi.org/10.1046/j.1523-1739.1998.96392.x

Richardson, D. M., & Higgins, S. I. (1998). Pines as invaders in the southern hemisphere. In D. M. Richardson & S. I. Higgins (Eds.), *Ecology and biogeography of Pinus*. Cambridge University Press.

Richardson, D. M., & Rejmanek, M. (2004). Conifers as invasive aliens: A global survey and predictive framework. *Diversity and Distributions*, *10*(5–6), Article 5–6. https://doi.org/10.1111/j.1366-9516.2004.00096.x

Richardson, S., Mill, A. C., Davis, D., Jam, D., & Ward, A. I. (2020). A systematic review of adaptive wildlife management for the control of invasive, non-native mammals, and other human–wildlife conflicts. *Mammal Review*, *50*(2), 147–156. https://doi.org/10.1111/mam.12182

Riley, S. (2012). Law is order, and good law is good order: The role of governance in the regulation of invasive alien species. *Environmental and Planning Law Journal*, *29*(1), 16–44. https://doi.org/10.3316/agispt.20121130

Rintoul, S. R., Chown, S. L., DeConto, R. M., England, M. H., Fricker, H. A., Masson-Delmotte, V., Naish, T. R., Siegert, M. J., & Xavier, J. C. (2018). Choosing the future of Antarctica. *Nature*, *558*, 233–241. https://doi.org/10.1038/s41586-018-0173-4

Roberts, S. (2003). Supply chain specific? Understanding the patchy success of ethical sourcing initiatives. In *Journal of business ethics* (Vol. 44, Issue 2, pp. 159–170). https://doi.org/10.1023/A:1023395631811

Robin, S., Wolcott, R., & Quintela, C. E. (2003). Perverse Subsidies and the Implications for Biodiversity: A review of recent findings and the status of policy reforms. *5th IUCN World Park Congress*, 8–17.

Robinson, C. J., Smyth, D., & Whitehead, P. J. (2005). Bush Tucker, Bush Pets, and Bush Threats: Cooperative Management of Feral Animals in Australia's Kakadu National Park. *Conservation Biology*, *19*(5), 1385–1391. https://doi.org/10.1111/j.1523-1739.2005.00196.x

Rocca-Serra, P., Salek, R. M., Arita, M., Correa, E., Dayalan, S., Gonzalez-Beltran, A., Ebbels, T., Goodacre, R., Hastings, J., Haug, K., Koulman, A., Nikolski, M., Oresic, M., Sansone, S.-A., Schober, D., Smith, J., Steinbeck, C., Viant, M. R., & Neumann, S. (2016). Data standards can boost metabolomics research, and if there is a will, there is a way. *Metabolomics*, *12*(1), 14. https://doi.org/10.1007/s11306-015-0879-3

Roder, W. (2001). *Slash-and-burn rice systems in the hills of Northern Lao PDR: Description, challenges, and opportunities*. International Rice Research Institute.

Rodríguez-Labajos, B., Binimelis, R., & Monterroso, I. (2009). Multi-level driving forces of biological invasions. *Ecological Economics*, *69*(1), 63–75. https://doi.org/10.1016/j.ecolecon.2009.08.022

Roman, J., & Darling, J. A. (2007). Paradox lost: Genetic diversity and the success of aquatic invasions. *Trends in Ecology & Evolution*, *22*(9), 454–464. https://doi.org/10.1016/j.tree.2007.07.002

Rondinelli, D. A., & Berry, M. A. (2000). Environmental citizenship in multinational corporations: Social responsibility and sustainable development. In *European Management Journal* (Vol. 18, Issue 1, pp. 70–84). https://doi.org/10.1016/S0263-2373(99)00070-5

Rosenthal, M. F., Gertler, M., Hamilton, A. D., Prasad, S., & Andrade, M. C. B. (2017). Taxonomic bias in animal behaviour publications. *Animal Behaviour*, *127*, 83–89. https://doi.org/10.1016/j.anbehav.2017.02.017

Rouillard, J., Lago, M., Abhold, K., Roeschel, L., Kafyeke, T., Klimmek, H., & Mattheiß, V. (2018). Protecting and Restoring Biodiversity across the Freshwater, Coastal and Marine Realms: Is the existing EU policy framework fit for purpose? *Environmental Policy and Governance*, *28*(2), 114–128. https://doi.org/10.1002/eet.1793

Roura-Pascual, N., Leung, B., Rabitsch, W., Rutting, L., Vervoort, J., Bacher, S., Dullinger, S., Erb, K.-H., Jeschke, J. M., Katsanevakis, S., Kühn, I., Lenzner, B., Liebhold, A. M., Obersteiner, M., Pauchard, A., Peterson, G. D., Roy, H. E., Seebens, H., Winter, M., … Essl, F. (2021). Alternative futures for global biological invasions. *Sustainability Science*, *16*(5), 1637–1650. https://doi.org/10.1007/s11625-021-00963-6

Roy, H. E., Groom, Q., Adriaens, T., Agnello, G., Antic, M., Archambeau, A.-S., Bacher, S., Bonn, A., Brown, P., Brundu, G., López, B. C., Cleary, M., Cogălniceanu, D., de Groot, M., De Sousa, T., Deidun, A., Essl, F., Fišer Pečnikar, Ž., Gazda, A., … Cardoso, A. C. (2018). Increasing understanding of alien species through citizen science (Alien-CSI). *Research Ideas and Outcomes*, *4*, e31412. https://doi.org/10.3897/rio.4.e31412

Roy, H. E., & Roy, D. B. (2008). *Deliverable: Review of invasive alien species monitoring in Europe including a plan for an early warning network.* DAISIE project deliverables.

Ruckelshaus, M. H., Jackson, S. T., Mooney, H. A., Jacobs, K. L., Kassam, K.-A. S., Arroyo, M. T. K., Báldi, A., Bartuska, A. M., Boyd, J., Joppa, L. N., Kovács-Hostyánszki, A., Parsons, J. P., Scholes, R. J., Shogren, J. F., & Ouyang, Z. (2020). The IPBES Global Assessment: Pathways to Action. *Trends in Ecology & Evolution*, *35*(5), 407–414. https://doi.org/10.1016/j.tree.2020.01.009

Rudd, M. A., Dickey-Collas, M., Ferretti, J., Johannesen, E., Macdonald, N. M., McLaughlin, R., Rae, M., Thiele, T., & Link, J. S. (2018). Ocean Ecosystem-Based Management Mandates and Implementation in the North Atlantic. *Frontiers in Marine Science*, *5*. https://doi.org/10.3389/fmars.2018.00485

Russell, J. C., Jones, H. P., Armstrong, D. P., Courchamp, F., Kappes, P. J., Seddon, P. J., Oppel, S., Rauzon, M. J., Cowan, P. E., Rocamora, G., Genovesi, P., Bonnaud, E., Keitt, B. S., Holmes, N. D., & Tershy, B. R. (2016). Importance of lethal control of invasive predators for island conservation. *Conservation Biology*, *30*(3), 670–672. https://doi.org/10.1111/cobi.12666

Ruttenberg, B. I., Schofield, P. J., Akins, J. L., Acosta, A., Feeley, M. W., Blondeau, J., Smith, S. G., & Ault, J. S. (2012). Rapid invasion of the Indo-Pacific lionfishes (*Pterois volitans* and *Pterois miles*) in the Florida Keys, USA: evidence from multiple pre-and post-invasion data sets. *Bulletin of Marine Science*, *88*(4), 10511059. https://doi.org/10.5343/bms.2011.1108

Sandmo, A. (1975). Optimal taxation in the presence of externalities. In *The Swedish Journal of Economics* (Vol. 77, Issue 1, pp. 86–98). https://doi.org/10.2307/3439329

Sandmo, A. (2008). Pigouvian taxes. In *The new Palgrave dictionary of economics* (Vol. 2). Palgrave Macmillan. https://doi.org/10.1057/978-1-349-95121-5_2678-1

Sarà, G., Palmeri, V., Rinaldi, A., Montalto, V., & Helmuth, B. (2013). Predicting biological invasions in marine habitats through eco-physiological mechanistic models: A case study with the bivalve *Brachidontes pharaonis*. *Diversity & Distributions*, *19*(10), 1235–1247. https://doi.org/10.1111/ddi.12074

Sarat, E., Dutartre, A., Soubeyran, Y., & Poulet, N. (2017). A French working group on biological invasions in aquatic environments: Towards an improvement of knowledge and management of freshwater invasive alien species. *Management of Biological Invasions*, *8*(3), 415–424. https://doi.org/10.3391/mbi.2017.8.3.15

Saul, W.-C., Roy, H. E., Booy, O., Carnevali, L., Chen, H.-J., Genovesi, P., Harrower, C. A., Hulme, P. E., Pagad, S., Pergl, J., & Jeschke, J. M. (2017). Assessing patterns in introduction pathways of alien species by linking major invasion data bases. *Journal of Applied Ecology*, *54*(2), 657–669. https://doi.org/10.1111/1365-2664.12819

SCAR & COMNAP. (2019). *Inter-continental Checklists: For supply chain managers of the National Antarctic Programmes for the reduction in risk of transfer of non-native species*. Scientific Committee on Antarctic Research & Council of Managers of National Antarctic Programs. https://documents.ats.aq/ATCM42/att/ATCM42_att068_e.pdf

Schofield, P. J. (2009). Geographic extent and chronology of the invasion of non-native lionfish (*Pterois volitans* [Linnaeus 1758] and *P. miles* [Bennett 1828]) in the Western North Atlantic and Caribbean Sea. *Aquatic Invasions*, *4*(3), 473–479. https://doi.org/10.3391/ai.2009.4.3.5

Schot, J., & Steinmueller, W. E. (2018). Three frames for innovation policy: R&D, systems of innovation and transformative change. *Research Policy*, *47*(9), 1554–1567. https://doi.org/10.1016/j.respol.2018.08.011

Schuster, R., Germain, R. R., Bennett, J. R., Reo, N. J., & Arcese, P. (2019). Vertebrate biodiversity on indigenous-managed lands in Australia, Brazil, and Canada equals that in protected areas. *Environmental Science & Policy*, *101*, 1–6. https://doi.org/10.1016/j.envsci.2019.07.002

Schwindt, E., & Bortolus, A. (2017). Aquatic invasion biology research in South America: Geographic patterns, advances and perspectives. *Aquatic Ecosystem Health & Management*, *20*(4), 322–333. https://doi.org/10.1080/14634988.2017.1404413

Scoones, I., Stirling, A., Abrol, D., Atela, J., Charli-Joseph, L., Eakin, H., Ely, A., Olsson, P., Pereira, L., Priya, R., van Zwanenberg, P., & Yang, L. (2020). Transformations to sustainability: Combining structural, systemic and enabling approaches. *Current Opinion in Environmental Sustainability*, *42*, 65–75. https://doi.org/10.1016/j.cosust.2019.12.004

SEAM. (2006). *Resolución N° 1.185/06—Por la cual se establecen los Términos Oficiales de Referencia para la introducción de especies exóticas de flora.* https://www.ecolex.org/details/legislation/resolucion-n-118506-por-la-cual-se-establecen-los-terminos-oficiales-de-referencia-para-la-introduccion-de-especies-exoticas-de-flora-lex-faoc087501/

Seaton, K., Bettin, A., & Grüneberg, H. (2014). New Ornamental Plants for Horticulture. In G. R. Dixon & D. E. Aldous (Eds.), *Horticulture: Plants for People and Places, Volume 1: Production Horticulture* (pp. 435–463). Springer Netherlands. https://doi.org/10.1007/978-94-017-8578-5_12

Secretariat, Great Britain Non-native Species. (2015). *The Great Britain Invasive Non-native Species Strategy*.

Secretariat of Convention on Wetlands. (1971). *Convention on Wetlands of International Importance especially as Waterfowl Habitat: Final Text adopted by the International Conference on the Wetlands and Waterfowl*. https:// www.ramsar.org/sites/default/files/ documents/library/original_1971_ convention_e.pdf

Secretariat of the CBD. (2001). *Review of the efficiency and efficacy of existing legal instruments applicable to invasive alien species* (CBD Technical Series no. 2; p. 42). Secretariat of the Convention on Biological Diversity.

Secretariat of the CBD. (2014). *Global Biodiversity Outlook 4* (p. 155). Secretariat of the Convention on Biological Diversity. https://www.cbd.int/gbo/gbo4/publication/gbo4-en-hr.pdf

Secretariat of the CBD. (2020). *Global Biodiversity Outlook 5* (p. 211). Secretariat of the Convention on Biological Diversity. https://www.cbd.int/gbo/gbo5/publication/gbo-5-en.pdf

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., van Kleunen, M., Winter, M., Ansong, M., Arianoutsou, M., Bacher, S., Blasius, B., Brockerhoff, E. G., Brundu, G., Capinha, C., Causton, C. E., Celesti-Grapow, L., … Essl, F. (2018). Global rise in emerging alien species results from increased accessibility of new source pools. *Proceedings of the National Academy of Sciences*, *115*(10), E2264–E2273. https://doi.org/10.1073/pnas.1719429115

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., … Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications*, *8*(1), 14435. https://doi.org/10.1038/ncomms14435

Seebens, H., Clarke, D. A., Groom, Q., Wilson, J. R. U., García-Berthou, E., Kühn, I., Roigé, M., Pagad, S., Essl, F., Vicente, J., Winter, M., & McGeoch, M. A. (2020). A workflow for standardising and integrating alien species distribution data. *NeoBiota*, *59*, 39–59. https://doi.org/10.3897/neobiota.59.53578

Seebens, H., Essl, F., Dawson, W., Fuentes, N., Moser, D., Pergl, J., Pyšek, P., Van Kleunen, M., Weber, E., Winter, M., & Blasius, B. (2015). Global trade will accelerate plant invasions in emerging economies under climate change. *Global Change Biology*, *21*(11), 4128–4140. https://doi.org/10.1111/gcb.13021

Segerson, K. (1988). Uncertainty and incentives for nonpoint pollution control. *Journal of Environmental Economics and Management*, *15*(1), 87–98. https://doi.org/10.1016/0095-0696(88)90030-7

Seipel, T., Kueffer, C., Rew, L. J., Daehler, C. C., Pauchard, A., Naylor, B. J., Alexander, J. M., Edwards, P. J., Parks, C. G., Arevalo, J. R., Cavieres, L. A., Dietz, H., Jakobs, G., McDougall, K., Otto, R., & Walsh, N. (2012). Processes at multiple scales affect richness and similarity of non-native plant species in mountains around the world. *Global Ecology and Biogeography*, *21*(2), 236–246. https://doi.org/10.1111/j.1466-8238.2011.00664.x

Sen, A. (1982). Approaches to the choice of discount rates for social cost benefit analysis. In R. C. Lind (Ed.), *Discounting for Time an Risk in Energy Policy*. Taylor & Francis Inc. https://scholar.harvard.edu/sen/publications/approaches-choice-discount-rates-social-cost-benefit-analysis

Sethi, S. P. (2011). Self-regulation through voluntary codes of conduct. In *Globalization and Self-Regulation* (pp. 3–14). Palgrave Macmillan. https://doi.org/10.1057/9780230348578_1

Shabani, F., Ahmadi, M., Kumar, L., Solhjouy-fard, S., Tehrany, M. S., Shabani, F., Kalantar, B., & Esmaeili, A. (2020). Invasive weed species' threats to global biodiversity: Future scenarios of changes in the number of invasive species in a changing climate. In *Ecological Indicators* (Vol. 116, p. 106436). https://doi.org/10.1016/j.ecolind.2020.106436

Shackleton, R. T., Adriaens, T., Brundu, G., Dehnen-Schmutz, K., Estévez, R. A., Fried, J., Larson, B. M. H., Liu, S., Marchante, E., Marchante, H., Moshobane, M. C., Novoa, A., Reed, M., & Richardson, D. M. (2019). Stakeholder engagement in the study and management of invasive alien species. *Journal of Environmental Management*, *229*, 88–101. https://doi.org/10.1016/j.jenvman.2018.04.044

Shackleton, R. T., Le Maitre, D. C., van Wilgen, B. W., & Richardson, D. M. (2015). The impact of invasive alien *Prosopis* species (mesquite) on native plants in different environments in South Africa. *South African Journal of Botany*, *97*, 25–31. https://doi.org/10.1016/j.sajb.2014.12.008

Shackleton, R. T., Richardson, D. M., Shackleton, C. M., Bennett, B., Crowley, S. L., Dehnen-Schmutz, K., Estévez, R. A., Fischer, A., Kueffer, C., Kull, C. A., Marchante, E., Novoa, A., Potgieter, L. J., Vaas, J., Vaz, A. S., & Larson, B. M. H. (2019). Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management*, *229*, 10–26. https://doi.org/10.1016/j.jenvman.2018.04.045

Shackleton, R. T., Shackleton, C. M., & Kull, C. A. (2019). The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management*, *229*, 145–157. https://doi.org/10.1016/j.jenvman.2018.05.007

Shackleton, R. T., Witt, A. B. R., Piroris, F. M., & van Wilgen, B. W. (2017). Distribution and socio-ecological impacts of the invasive alien cactus *Opuntia stricta* in eastern Africa. *Biological Invasions*, *19*(8), 2427–2441. https://doi.org/10.1007/s10530-017-1453-x

Shanungu, G. K. (2009). Management of the invasive *Mimosa pigra* L. in Lochinvar National Park, Zambia. *Biodiversity*, *10*(2–3), 56–60. https://doi.org/10.1080/14888386.2009.9712844

Shine, C. (2003). Overview of international instruments addressing invasive alien species. In N. Pallewatta, J. Reaser, & A. Gutierrez (Eds.), *Prevention and Management of Invasive Alien Species: Proceedings of a Workshop on Forging Cooperation throughout South and Southeast Asia* (pp. 19–35). Global Invasive Species Programme. https://www.researchgate.net/publication/303442790_Prevention_and_Management_of_Invasive_Alien_Species_Proceedings_of_a_Workshop_on_Forging_Cooperation_throughout_South_and_Southeast_Asia

Shine, C. (2007). Invasive species in an international context: IPPC, CBD, European Strategy on Invasive Alien Species and other legal instruments. *EPPO Bulletin*, *37*(1), 103–113. https://doi.org/10.1111/j.1365-2338.2007.01087.x

Shine, C., Williams, N., & Gündling, L. (2000). *A guide to designing legal and institutional frameworks on alien invasive species*. IUCN. https://www.nobanis.org/globalassets/articlesreports/a-guide-to-designing-legal-framework-on-ias.pdf

Shine, C., Williams, N. M., & Burnhenne-Guilmin, F. (2005). Legal and institutional frameworks for invasive alien species. In H. A. Mooney, R. N. Mack, J. A. McNeely, L. E. Neville, P. J. Schei, & J. K. Waage (Eds.), *Invasive alien species: A new synthesis* (pp. 233–284). Island Press.

Shrestha, B. B., Shrestha, U. B., Sharma, K. P., Thapa-Parajuli, R. B., Devkota, A., & Siwakoti, M. (2019). Community perception and prioritization of invasive alien plants in Chitwan-Annapurna Landscape, Nepal. *Journal of Environmental Management*, *229*, 38–47. https://doi.org/10.1016/j.jenvman.2018.06.034

Sillitoe, P. (1998). The development of indigenous knowledge: A new applied anthropology. *Current Anthropology*, *39*(2), 223–252. https://doi.org/10.1086/204722

Sillitoe, P., & Marzano, M. (2009). Future of indigenous knowledge research in development. *Futures*, *41*(1), 13–23. https://doi.org/10.1016/j.futures.2008.07.004

Simberloff, D. (2021). Maintenance management and eradication of established aquatic invaders. *Hydrobiologia*, *848*(9), 2399–2420. https://doi.org/10.1007/s10750-020-04352-5

Simberloff, D., Nuñez, M. A., Ledgard, N. J., Pauchard, A., Richardson, D. M., Sarasola, M., Van Wilgen, B. W., Zalba, S. M., Zenni, R. D., Bustamante, R., Peña, E., & Ziller, S. R. (2009). Spread and impact of introduced conifers in South America: Lessons from other southern hemisphere regions. *Austral Ecology*, *35*(5), 489–504. https://doi.org/10.1111/j.1442-9993.2009.02058.x

Singh, R. S., & Kaur, J. (2002). Alien plant pathogens in India. *Biological Invasions*, 159–170.

Skultety, D., & Matthews, J. W. (2017). Urbanization and roads drive non-native plant invasion in the Chicago Metropolitan region. *Biological Invasions*, *19*(9), 2553–2566. https://doi.org/10.1007/s10530-017-1464-7

Smith, A. L., Klenk, N., Wood, S., Hewitt, N., Henriques, I., Yan, N., & Bazely, D. R. (2013). Second generation biofuels and bioinvasions: An evaluation of invasive risks and policy responses in the United States and Canada. *Renewable and Sustainable Energy Reviews*, *27*, 30–42. https://doi.org/10.1016/j.rser.2013.06.013

Smith, D., Hinz, H., Mulema, J., Weyl, P., & Ryan, M. J. (2018). Biological control and the Nagoya Protocol on access and benefit sharing–a case of effective due diligence. *Biocontrol Science and Technology*, *28*(10), 914–926. https://doi.org/10.1080/09583157.2018.1460317

Smolarz, K., Biskup, P., & Zgrundo, A. (2016). Biological Invasions: A Case Study of Baltic Sea Environmental Governance. In M. Gilek, M. Karlsson, S. Linke, & K. Smolarz (Eds.), *Environmental Governance of the Baltic Sea* (Vol. 10, pp. 73–95). Springer. https://doi.org/10.1007/978-3-319-27006-7_4

Söderström, S., Kern, K., Boström, M., & Gilek, M. (2016). "Environmental governance" and "ecosystem management": Avenues for synergies between two approaches. *Interdisciplinary Environmental Review*, *17*(1), 1–19. https://doi.org/10.1504/IER.2016.074871

Solomon Genet, B. (2007). *Shrub encroachment into grassland and its impact on Kafue Lechwe in Lochinvar National Park, Zambia* [Master of Science in Geo-Information Science and Earth Observation, University of Twente International Institute for Geo-Information Science and Earth Observation (ITC)]. https://doi.org/10.13140/RG.2.1.4126.7287

Soroye, P., Ahmed, N., & Kerr, J. T. (2018). Opportunistic citizen science data transform understanding of species distributions, phenology, and diversity gradients for global change research. *Global Change Biology*, *24*(11), 5281–5291. https://doi.org/10.1111/gcb.14358

Sovacool, B. K. (2017). Reviewing, Reforming, and Rethinking Global Energy Subsidies: Towards a Political Economy Research Agenda. *Ecological Economics*, *135*, 150–163. https://doi.org/10.1016/j.ecolecon.2016.12.009

Spencer, R. D., & Cross, R. G. (2007). The international code of botanical nomenclature (ICBN), the international code of nomenclature for cultivated plants (ICNCP), and the cultigen. *Taxon*, *56*(3), 938–940.

Speziale, K. L., Lambertucci, S. A., Carrete, M., & Tella, J. L. (2012). Dealing with non-native species: What makes the difference in South America? *Biological Invasions*, *14*(8), 1609–1621. https://doi.org/10.1007/s10530-011-0162-0

Springborn, M., Romagosa, C. M., & Keller, R. P. (2011). The value of nonindigenous species risk assessment in international trade. *Ecological Economics*, *70*(11), Article 11. https://doi.org/10.1016/j.ecolecon.2011.06.016

Staddon, S. C., Nightingale, A., & Shrestha, S. K. (2015). Exploring participation in ecological monitoring in Nepal's community forests. *Environmental Conservation*, *42*(3), 268–277. https://doi.org/10.1017/S037689291500003X

Stallman, H. R., & James, H. S. (2015). Determinants affecting farmers' willingness to cooperate to control pests. *Ecological Economics*, *117*, 182–192. https://doi.org/10.1016/j.ecolecon.2015.07.006

Staples, D. J., & Hermes, R. (2012). Marine biodiversity and resource management—What is the link? *Aquatic Ecosystem Health & Management*, *15*(3), 245–252. https://doi.org/10.1080/14634988.2012.709429

Stilgoe, J., Owen, R., & Macnaghten, P. (2020). Developing a framework for responsible innovation. *Research Policy*, *42*(9), 1568–1580. https://doi.org/10.1016/j.respol.2013.05.008

Stilwell, M., & Turk, E. (1999). Trade measures and multilateral environmental agreements: Resolving uncertainty and removing the WTO chill factor. *A WWF International Discussion Paper*. https://wwfint.awsassets.panda.org/downloads/chillfactor.pdf

Stirling, A. (2010). Keep it complex. *Nature*, *468*(7327), Article 7327. https://doi.org/10.1038/4681029a

Stockhammer, E., Hochreiter, H., Obermayr, B., & Steiner, K. (1997). The index of sustainable economic welfare (ISEW) as an alternative to GDP in measuring economic welfare. The results of the Austrian (revised) ISEW calculation 1955–1992. *Ecological Economics*, *21*(1), 19–34. https://doi.org/10.1016/S0921-8009(96)00088-2

Stoett, P. (2007). Counter-bioinvasion: Conceptual and governance challenges. *Environmental Politics*, *16*(3), 433–452. https://doi.org/10.1080/09644010701251672

Stoett, P. (2010). Framing Bioinvasion: Biodiversity, Climate Change, Security, Trade, and Global Governance. *Global Governance: A Review of Multilateralism and International Organizations*, *16*(1), 103–120. https://doi.org/10.1163/19426720-01601007

Stoett, P., Roy, H. E., & Pauchard, A. (2019). Invasive alien species and planetary and global health policy. *The Lancet Planetary Health*, *3*(10), e400–e401. https://doi.org/10.1016/S2542-5196(19)30194-9

Stokes, K. E., O°Neill, K. P., Montgomery, W. I., Dick, J. T. A., Maggs, C. A., & McDonald, R. A. (2006). The Importance of Stakeholder Engagement in Invasive Species Management: A Cross-jurisdictional Perspective in Ireland. *Biodiversity and Conservation*, *15*(8), 2829–2852. https://doi.org/10.1007/s10531-005-3137-6

Stone, C. D. (2004). Common but differentiated responsibilities in international law. In *American Journal of International Law* (Vol. 98, Issue 2, pp. 276–301). https://doi.org/10.2307/3176729

Suiter, K., & Sferrazza, S. (2007). Monitoring the sale and trafficking of invasive vertebrate species using automated internet search and surveillance tools. *Managing Vertebrate Invasive Species*, *51*. https://digitalcommons.unl.edu/nwrcinvasive/51

Sullivan, A., York, A. M., White, D. D., Hall, S. J., & Yabiku, S. T. (2017). *De jure versus de facto* institutions: Trust, information, and collective efforts to manage the invasive mile-a-minute weed (*Mikania micrantha*). *The Commons Journal*, *11*(1), 171–199. https://doi.org/10.18352/ijc.676

Sullivan, B. L., Aycrigg, J. L., Barry, J. H., Bonney, R. E., Bruns, N., Cooper, C. B., Damoulas, T., Dhondt, A. A., Dietterich, T., Farnsworth, A., Fink, D., Fitzpatrick, J. W., Fredericks, T., Gerbracht, J., Gomes, C., Hochachka, W. M., Iliff, M. J., Lagoze, C., La Sorte, F. A., … Kelling, S. (2014). The eBird enterprise: An integrated approach to development and application of citizen science. *Biological Conservation*, *169*, 31–40. https://doi.org/10.1016/j.biocon.2013.11.003

Sustainable Development Solution Network. (2021). *Indicators and a Monitoring Framework Launching a data revolution for the Sustainable Development Goals*. https://indicators.report/targets/15-8/

Swart, N. C., Gille, S. T., Fyfe, J. C., & Gillett, N. P. (2018). Recent Southern Ocean warming and freshening driven by greenhouse gas emissions and ozone depletion. *Nature Geoscience*, *11*, 836–841. https://doi.org/10.1038/s41561-018-0226-1

Taleb, N. N. (2007). *The black swan: The impact of the highly improbable*. Random House. http://catalog.hathitrust.org/api/volumes/oclc/71833470.html

Taylor, R. V., Holthuijzen, W., Humphrey, A., & Posthumus, E. (2020). Using phenology data to improve control of invasive plant species: A case study on Midway Atoll NWR. *Ecological Solutions and Evidence*, *1*(1), e12007. https://doi.org/10.1002/2688-8319.12007

Tekiela, D. R., & Barney, J. N. (2017). Invasion Shadows: The Accumulation and Loss of Ecological Impacts from an Invasive Plant. *Invasive Plant Science and Management*, *10*(1), 1–8. https://doi.org/10.1017/inp.2017.3

Temper, L., Walter, M., Rodriguez, I., Kothari, A., & Turhan, E. (2018). A perspective on radical transformations to sustainability: Resistances, movements and alternatives. *Sustainability Science*, *13*(3), 747–764. https://doi.org/10.1007/s11625-018-0543-8

Termeer, C. J. A. M., & Dewulf, A. (2014). Scale-Sensitivity as a Governance Capability: Observing, Acting and Enabling. In F. Padt, P. Opdam, N. Polman, & C. J. A. M. Termeer (Eds.), *Scale-sensitive Governance of the Environment* (pp. 38–55). John Wiley & Sons, Ltd. https://doi.org/10.1002/9781118567135.ch3

Termeer, C. J. A. M., Dewulf, A., Karlsson-Vinkhuyzen, S. I., Vink, M., & van Vliet, M. (2016). Coping with the wicked problem of climate adaptation across scales: The Five R Governance Capabilities. *Landscape and Urban Planning*, *154*, 11–19. http://dx.doi.org/10.1016/j.landurbplan.2016.01.007

Termeer, C. J. A. M., Dewulf, A., & Van Lieshout, M. (2010). Disentangling scale approaches in governance research: Comparing monocentric, multilevel, and adaptive governance. *Ecology and Society*, *15*(4), 29. http://www.ecologyandsociety.org/vol15/iss4/art29/

Theobald, E. J., Ettinger, A. K., Burgess, H. K., DeBey, L. B., Schmidt, N. R., Froehlich, H. E., Wagner, C., HilleRisLambers, J., Tewksbury, J., Harsch, M. A., & Parrish, J. K. (2015). Global change and local solutions: Tapping the unrealized potential of citizen science for biodiversity research. *Biological Conservation*, *181*, 236–244. https://doi.org/10.1016/j.biocon.2014.10.021

Thiel, M., Penna-Díaz, M. A., Luna-Jorquera, G., Salas, S., Sellanes, J., & Stotz, W. (2014). Citizen scientist and marine research: Volunteers participatns, their contribbutions, and projection for the future. In R. N. Hughes, D. J. Hughes, & I. P. Smith (Eds.), *Oceanography and Marine Biology: An annual review* (Vol. 52, p. 52). CRC Press. http://dx.doi.org/10.1201/b17143-6

Thomas, I. J. (2007). *Mapping and modelling of* Mimosa pigra *expansion in Lochinvar National Park, Zambia* [Master of Science in Geo-Information Science and Earth Observation, International Institute for Geo-information Science and Earth Observation]. https://webapps.itc.utwente.nl/librarywww/papers_2007/msc/gem/indira.pdf

Tiago, P., Pereira, H. M., & Capinha, C. (2017). Using citizen science data to estimate climatic niches and species distributions. *Basic and Applied Ecology*, *20*, 75–85. https://doi.org/10.1016/j.baae.2017.04.001

Tilahun, M., Birner, R., & Ilukor, J. (2017). Household-level preferences for mitigation of *Prosopis juliflora* invasion in the Afar region of Ethiopia: A contingent valuation. *Journal of Environmental Planning and Management*, *60*(2), 282–308. https://doi.org/10.1080/09640568.2016.1152955

Timus, N., & Babutsidze, Z. (2016). Pirating European Studies. *Journal of Contemporary European Research*, *12*(3), 783–791. https://www.proquest.com/docview/2673249500?pq-origsite=gscholar&fromopenview=true

Tollington, S., Turbé, A., Rabitsch, W., Groombridge, J. J., Scalera, R., Essl, F., & Shwartz, A. (2017). Making the EU Legislation on Invasive Species a Conservation Success. *Conservation Letters*, *10*(1), 112–120. https://doi.org/10.1111/conl.12214

Touza, J., Pérez-Alonso, A., Chas-Amil, M. L., & Dehnen-Schmutz, K. (2014). Explaining the rank order of invasive plants by stakeholder groups. *Ecological Economics*, *105*, 330–341. https://doi.org/10.1016/j.ecolecon.2014.06.019

Trégarot, E., Fumaroli, M., Arqué, A., Hellio, C., & Maréchal, J.-P. (2015). First records of the red lionfish (*Pterois volitans*) in Martinique, French West Indies: Monitoring invasion status through visual surveys. *Marine Biodiversity Records*, *8*, e1. https://doi.org/10.1017/S1755267214001341

Trillo, T. (2021). Using Multilateral Development Banks to Achieve Paris Climate Goals. *Environmental Law Reporter*, *51*(9), 10762–10772. https://heinonline.org/HOL/Page?handle=hein.journals/elrna51&id=1&collection=journals&index=

Troudet, J., Grandcolas, P., Blin, A., Vignes-Lebbe, R., & Legendre, F. (2017). Taxonomic bias in biodiversity data and societal preferences. *Scientific Reports*, *7*(1), 9132. https://doi.org/10.1038/s41598-017-09084-6

Trump, B. D., Foran, C., Rycroft, T., Wood, M. D., Bandolin, N., Cains, M., Cary, T., Crocker, F., Friedenberg, N. A., Gurian, P., Hamilton, K., Hoover, J. J., Meyer, C., Pokrzywinski, K., Ritterson, R., Schulte, P., Warner, C., Perkins, E., & Linkov, I. (2018). Development of community of practice to support quantitative risk assessment for synthetic biology products: Contaminant bioremediation and invasive carp control as cases. *Environment Systems & Decisions*, *38*(4), 517–527. https://doi.org/10.1007/s10669-018-9710-9

Turbelin, A. J., Malamud, B. D., & Francis, R. A. (2017). Mapping the global state of invasive alien species: Patterns of invasion and policy responses. *Global Ecology and Biogeography*, *26*(1), 78–92. https://doi.org/10.1111/geb.12517

Turchini, G. M., & De Silva, S. S. (2008). Bio-economical and ethical impacts of alien finfish culture in European inland waters. *Aquaculture International*, *16*(3), 243–272. https://doi.org/10.1007/s10499-007-9141-y

Turnhout, E., Metze, T., Wyborn, C., Klenk, N., & Louder, E. (2020). The politics of co-production: Participation, power, and transformation. *Current Opinion in Environmental Sustainability*, *42*, 15–21. https://doi.org/10.1016/j.cosust.2019.11.009

Tye, A. (Ed.). (2018). *Guidelines for invasive species planning and management on islands* (1st ed.). IUCN, International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2018.15.en

Udovyk, O., & Gilek, M. (2013). Coping with uncertainties in science-based advice informing environmental management of the Baltic Sea. *Environmental Science & Policy*, *29*, 12–23. https://doi.org/10.1016/j.envsci.2013.01.015

UK Parliament. (2019). *Invasive species—Environmental Audit Committee—House of Commons*. https://publications.parliament.uk/pa/cm201919/cmselect/cmenvaud/88/8804.htm

UN. (2008). *Reforming energy subsidies: Opportunities to contribute to the climate change agenda*. United Nations Environment Programme. https://wedocs.unep.org/20.500.11822/25941

UN. (2021). *SDG Indicators—Metadata repository* [United Nations Statistics Division, Development Data and Outreach Branch]. https://unstats.un.org/sdgs/metadata/?Text=&Goal=15&Target=15.8

Underwood, E., Scalera, R., & Rabitsch, W. (2020). *FAQs on how the EU Invasive Alien Species Regulation interacts with the EU Nature Directives*. https://ieep.eu/publications/faqs-on-how-the-eu-invasive-alien-species-regulation-interacts-with-the-eu-nature-directives/

UNEP. (2014). *Action Plan concerning species introductions and invasive species in the Mediterranean Sea*. UNDP. https://www.undp.org/turkiye/publications/action-plan-concerning-species-introductions-and-invasive-species-mediterranean-sea

UNESCO. (2017). *The World Heritage Convention*. The United Nations Educational, Scientific and Cultural Organization. http://whc.unesco.org/en/ Convention/

Ura, K., Alkire, S., Zangmo, T., & Wangdi, K. (2012). *A short guide to gross national happiness index*. https://www.gov.uk/research-for-development-outputs/a-short-guide-to-gross-national-happiness-index

Usher, M. B., Kruger, F. J., Macdonald, I. A. W., Loope, L. L., & Brockie, R. E. (1988). The Ecology of Biological Invasions into Nature Reserves—An Introduction. *Biological Conservation*, *44*(1–2), 1–8. https://doi.org/10.1016/0006-3207(88)90002-X

Vaas, J., Driessen, P. P. J., Giezen, M., van Laerhoven, F., & Wassen, M. J. (2017). Who's in charge here anyway? Polycentric governance configurations and the development of policy on invasive alien species in the semisovereign Caribbean. *Ecology and Society*, *22*(4), 1. https://doi.org/10.5751/ES-09487-220401

van der Ven, H., Rothacker, C., & Cashore, B. (2018). Do eco-labels prevent deforestation? Lessons from non-state market driven governance in the soy, palm oil, and cocoa sectors. *Global Environmental Change*, *52*, 141–151. https://doi.org/10.1016/j.gloenvcha.2018.07.002

van Heezik, Y. M., Dickinson, K. J. M., & Freeman, C. (2012). Closing the Gap: Communicating to Change Gardening Practices in Support of Native Biodiversity in Urban Private Gardens. *Ecology and Society*, *17*(1), Article 1. https://doi.org/10.5751/Es-04712-170134

van Rees, C. B., Hand, B. K., Carter, S. C., Bargeron, C., Cline, T. J., Daniel, W., Ferrante, J. A., Gaddis, K., Hunter, M. E., Jarnevich, C. S., McGeoch, M. A., Morisette, J. T., Neilson, M. E., Roy, H. E., Rozance, M. A., Sepulveda, A., Wallace, R. D., Whited, D., Wilcox, T., … Luikart, G. (2022). A framework to integrate innovations in invasion science for proactive management. *Biological Reviews*, *97*(4), 1712–1735. https://doi.org/10.1111/brv.12859

Vane, M., & Runhaar, H. A. C. (2016). Public support for invasive alien species eradication programs: Insights from the Netherlands. *Restoration Ecology*, *24*(6), 743–748. https://doi.org/10.1111/rec.12379

VanNijnatten, D. L. (2016). The Aquatic Invasion: Assembling Transboundary Governance Capacity for Prevention and Detection. *International Journal of Water Governance*, *4*(6), 91–110. https://doi.org/10.7564/15-ijwg84

Vaz, A. S., Kueffer, C., Kull, C. A., Richardson, D. M., Schindler, S., Muñoz-Pajares, A. J., Vicente, J. R., Martins, J., Hui, C., Kühn, I., & Honrado, J. P. (2017). The progress of interdisciplinarity in invasion science. *Ambio*, *46*(4), 428–442. https://doi.org/10.1007/s13280-017-0897-7

Verbrugge, L. N. H., Leuven, R. S. E. W., van Valkenburg, J. L. C. H., & van den Born, R. J. G. (2014). Evaluating stakeholder awareness and involvement in risk prevention of aquatic invasive plant species by a national code of conduct. *Aquatic Invasions*, *9*(3), 369–381. https://doi.org/10.3391/ai.2014.9.3.11

Vicente, J. R. S., Vaz, A. S., Roige, M., Winter, M., Clarke, D., Lenzner, B., & McGeoch, M. A. (2022). Existing indicators do not adequately monitor progress towards meeting invasive alien species targets. In *Conservation Letters* (Vol. 15, Issue 5, p. e12918). https://doi.org/10.1111/conl.12918

Victorian Government. (2010). *Invasive plants and animals: Policy framework*. Dept. of Primary Industries. https://agriculture.vic.gov.au/__data/assets/pdf_file/0009/582255/Invasive-Plants-and-Animals-Policy-Framework-IPAPF.pdf

Vines, T. H., Albert, A. Y. K., Andrew, R. L., Débarre, F., Bock, D. G., Franklin, M. T., Gilbert, K. J., Moore, J.-S., Renaut, S., & Rennison, D. J. (2014). The Availability of Research Data Declines Rapidly with Article Age. *Current Biology*, *24*(1), 94–97. https://doi.org/10.1016/j.cub.2013.11.014

Visseren-Hamakers, I. J. (2015). Integrative environmental governance: Enhancing governance in the era of synergies. *Current Opinion in Environmental Sustainability*, *14*, 136–143. https://doi.org/10.1016/j.cosust.2015.05.008

Visseren-Hamakers, I. J., Razzaque, J., McElwee, P., Turnhout, E., Kelemen, E., Rusch, G. M., Fernández-Llamazares, Á., Chan, I., Lim, M., Islar, M., Gautam, A. P., Williams, M., Mungatana, E., Karim, M. S., Muradian, R., Gerber, L. R., Lui, G., Liu, J., Spangenberg, J. H., & Zaleski, D. (2021). Transformative governance of biodiversity: Insights for sustainable development. *Current Opinion in Environmental Sustainability*, *53*, 20–28. https://doi.org/10.1016/j.cosust.2021.06.002

Wald, D. M., Nelson, K. A., Gawel, A. M., & Rogers, H. S. (2019). The role of trust in public attitudes toward invasive species management on Guam: A case study. *Journal of Environmental Management*, *229*, 133–144. https://doi.org/10.1016/j.jenvman.2018.06.047

Walker, K. R., Ricciardone, M. D., & Jensen, J. (2003). Developing an international consensus on DDT: a balance of environmental protection and disease control. *International Journal of Hygiene and Environmental Health*, *206*(4–5), Article 4–5. https://doi.org/10.1078/1438-4639-00239

Wallace, R. D., & Bargeron, C. T. (2014). Identifying invasive species in real time: Early detection and distribution mapping system (EDDMapS) and other mapping tools. In *Invasive Species and Global Climate Change* (Vol. 4, pp. 219–231). https://www.scopus.com/inward/record.uri?eid=2-s2.0-84927795231&partnerID=40&md5=b5a281e3a1844e2fdb58a863e3e3fcbe

Walters, L. J., Brown, K. R., Stam, W. T., & Olsen, J. L. (2006). E-commerce and *Caulerpa*: Unregulated dispersal of invasive species. *Frontiers in Ecology and the Environment*, *4*(2), 75–79. https://doi.org/10.1890/1540-9295(2006)004[0075:EACUDO]2.0.CO;2

Warwick, H. (2012). Comment: Uist Hedgehogs—Lessons learnt in wildlife management. *British Wildlife*, *24*(2), 111.

WCO. (2018). *WCO Cross-border e-commerce framework of standards*. World Customs Organization. http://www.wcoomd.org/-/media/wco/public/global/pdf/topics/facilitation/activities-and-programmes/ecommerce/wco-framework-of-standards-on-crossborder-ecommerce_en.pdf?la=en

WCO. (2019). *Technical Specifications Framework Of Standards On Cross-Border E-Commerce*. World Customs Organization. https://img1.wsimg.com/blobby/go/3868ba55-a36a-45cf-a1be-6d257a2f1a32/downloads/Technical%20Specifications%20Framework%20of%20Standard.pdf?ver=1594066770640

Weitz, N., Strambo, C., Kemp-Benedict, E., & Nilsson, M. (2017). Closing the governance gaps in the water-energy-food nexus: Insights from integrative governance. *Global Environmental Change*, *45*, 165–173. https://doi.org/10.1016/j.gloenvcha.2017.06.006

Westley, F., Olsson, P., Folke, C., Homer-Dixon, T., Vredenburg, H., Loorbach, D., Thompson, J., Nilsson, M., Lambin, E., Sendzimir, J., Banerjee, B., Galaz, V., & van der Leeuw, S. (2011). Tipping toward sustainability: Emerging pathways of transformation. *Ambio*, *40*(7), 762–780. https://doi.org/10.1007/s13280-011-0186-9

Westphal, M. I., Browne, M., MacKinnon, K., & Noble, I. (2008). The link between international trade and the global distribution of invasive alien species. *Biological Invasions*, *10*(4), 391–398. https://doi.org/10.1007/s10530-007-9138-5

Wilkinson, M. D., Dumontier, M., Aalbersberg, Ij. J., Appleton, G., Axton, M., Baak, A., Blomberg, N., Boiten, J.-W., da Silva Santos, L. B., Bourne, P. E., Bouwman, J., Brookes, A. J., Clark, T., Crosas, M., Dillo, I., Dumon, O., Edmunds, S., Evelo, C. T., Finkers, R., … Mons, B. (2016). The FAIR Guiding Principles for scientific data management and stewardship. *Scientific Data*, *3*(1), 160018. https://doi.org/10.1038/sdata.2016.18

Williams, T., & Hardison, P. (2013). Culture, law, risk and governance: Contexts of traditional knowledge in climate change adaptation. In J. K. Maldonado, B. Colombi, & R. Pandya (Eds.), *Climate Change and Indigenous Peoples in the United States* (pp. 23–36). Springer. https://doi.org/10.1007/978-3-319-05266-3_3

Wilson, E. O. (1999). *The diversity of life*. WW Norton & Company.

Wilson, J. R. U., Faulkner, K. T., Rahlao, S. J., Richardson, D. M., Zengeya, T. A., & van Wilgen, B. W. (2018). Indicators for monitoring biological invasions at a national level. *Journal of Applied Ecology*, *55*(6), 2612–2620. https://doi.org/10.1111/1365-2664.13251

Wise, L. K. (2019). iMapInvasives 3.0: New features and improvements. *Portland/Vancouver Urban Ecology & Conservation Symposium 2019: 17th Annual*. 2019 Urban Ecology & Conservation Symposium. https://archives.pdx.edu/ds/psu/33784

WOAH. (2011). *Guidelines For Assessing The Risk Of Non-Native Animals Becoming Invasive*. World Organization for Animal Health. https://www.woah.org/app/uploads/2021/03/oieguidelines-nonnativeanimals-2012.pdf

Wonham, M. J., & Carlton, J. T. (2005). Trends in marine biological invasions at local and regional scales: The Northeast Pacific Ocean as a model system. *Biological Invasions*, *7*(3), 369–392. https://doi.org/10.1007/s10530-004-2581-7

Woodford, D. J., Richardson, D. M., MacIsaac, H. J., Mandrak, N. E., van Wilgen, B. W., Wilson, J. R. U., & Weyl, O. L. F. (2016). Confronting the wicked problem of managing biological invasions. *NeoBiota*, *31*, 63–86. https://doi.org/10.3897/neobiota.31.10038

Wright, E. M., Bhammar, H. M., Gonzalez Velosa, A. M., & Sobrevila, C. (2016). *Analysis of international funding to tackle illegal wildlife trade*. The World Bank. https://documents1.worldbank.org/curated/en/695451479221164739/pdf/110267-WP-Illegal-Wildlife-Trade-OUO-9.pdf

WTO. (1995). *The WTO Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement)* (pp. 69–83). World Trade Organization. https://www.wto.org/english/docs_e/legal_e/15sps_01_e.htm

Yesson, C., Brewer, P. W., Sutton, T., Caithness, N., Pahwa, J. S., Burgess, M., Gray, W. A., White, R. J., Jones, A. C., Bisby, F. A., & Culham, A. (2007). How Global Is the Global Biodiversity Information Facility? *PLoS ONE*, *2*(11), e1124. https://doi.org/10.1371/journal.pone.0001124

Yung, L., Chandler, J., & Haverhals, M. (2015). Effective Weed Management, Collective Action, and Landownership Change in Western Montana. *Invasive Plant Science and Management*, *8*(2), 193–202. https://doi.org/10.1614/IPSM-D-14-00059.1

Zalom, F., Grieshop, J., Lelea, M. A., & Sedell, J. K. (2013). *Community perceptions of emergency responses to invasive species in California: Case studies of the light brown apple moth and the European grapevine moth* (pp. 10–8100) [Report submitted to USDA in fulfillment of Cooperative Agreement]. https://ucanr.edu/blogs/strawberries_caneberries/blogfiles/16221.pdf

Zamora-Gutierrez, V., Amano, T., & Jones, K. E. (2019). Spatial and taxonomic biases in bat records: Drivers and conservation implications in a megadiverse country. *Ecology and Evolution*, *9*(24), 14130–14141. https://doi.org/10.1002/ece3.5848

Zeng, H., Liu, X., Zhang, L., Li, Y., Zhu, M., & Chen, D. (2021). Educational approaches help bridge perception gaps of invasive alien species (*Mikania micrantha*) between managers and non-managers. *Environmental Management*, *68*, 340–352. https://doi.org/10.1007/s00267-021-01505-7

Zengeya, T. A., Ivey, P., Woodford, D. J., Weyl, O., Novoa, A., Shackleton, R., Richardson, D., & Van Wilgen, B. (2017). Managing conflict-generating invasive species in South Africa: Challenges and trade-offs. *Bothalia*, *47*(2), a2160. https://doi.org/10.4102/abc.v47i2.2160

Zengeya, T. A., & Wilson, J. R. (2020). *The status of biological invasions and their management in South Africa in 2019* (Final Report) [Final Report]. Zenodo. https://doi.org/10.5281/ZENODO.3947613

Zenni, R. D., & Nuñez, M. A. (2013). The elephant in the room: The role of failed invasions in understanding invasion biology. *Oikos*, *122*(6), 801–815. https://doi.org/10.1111/j.1600-0706.2012.00254.x

Zhao, Y.-J., Hosoya, T., Baral, H.-O., Hosaka, K., & Kakishima, M. (2013). *Hymenoscyphus pseudoalbidus*, the correct name for *Lambertella albida* reported from Japan. *Mycotaxon*, *122*(1), 25–41. https://doi.org/10.5248/122.25

ANNEXES

ANNEX I
Glossary

A

Adaptive governance: refers to flexible and learning-based collaborations and decision-making processes involving both state and nonstate actors, often at multiple levels, with the aim to adaptively negotiate and coordinate management of social–ecological systems and ecosystem services across landscapes and seascapes (Folke *et al.,* 2005).

Adaptive management: a philosophy that accepts that management must proceed even without complete information. It views management not only as a way to achieve objectives, but also as a process for probing to learn more about the resource or system being managed. Learning is an inherent objective of adaptive management. Adaptive management is a process where policies and activities can adapt to future conditions to improve management success (CCBA, 2008).

Alien species: a species whose presence in a region is attributable to human actions, intentional or unintentional, that enable them to overcome biogeographical barriers (Richardson *et al.,* 2010; Figure 1.1). This includes species, subspecies or lower taxon, and any part (gametes, seeds, eggs, or propagules) of such species that might survive and subsequently reproduce (CBD, 2002).

B

Biocultural community protocol: a biocultural community protocol is a document that is developed after a community under-takes a consultative process to outline their core cultural and spiritual values and customary laws relating to their traditional knowledge and resources (LPP and LIFE Network *et al.,* 2010).

Biocultural management (or biocultural approaches to conservation or biocultural approaches to environmental management): actions made in the service of sustaining the biophysical and sociocultural components of dynamic, interacting, and interdependent social–ecological systems (Gavin *et al.,* 2015; Lyver *et al.,* 2019).

Biological control: the use of living organisms to suppress the population density or impact of a specific invasive alien species, making it less abundant or less damaging than it would otherwise be (Eilenberg *et al.,* 2001).

Biological invasion (or invasion process): a process involving the transport of a native species outside of its natural range, intentionally or unintentionally, by human activities to new regions where it may become established, spread and ultimately adversely impact nature, nature's contributions to people, and good quality of life (Blackburn *et al.,* 2011; Figure 1.6).

Biosecurity: for the purpose of this assessment, a strategic and integrated approach that encompasses the policy and regulatory frameworks (including instruments and activities) for identifying, analysing and managing risks, including invasive alien species, to human, animal and plant life and health, and associated risks to the economy and the environment (FAO, 2007).

Biotic facilitation: any interaction where the action of one species has a beneficial effect on another. This includes mutualistic interactions where both the facilitated and facilitator benefit (+/+), those which are commensal (+/0) when the effects of the facilitated on the facilitator are neutral as well as those which are antagonistic (+/–) when the facilitated negatively impact the facilitator. Note that this concept partially overlaps with that of mutualism, ecological engineering and niche construction (Zélé *et al.,* 2018).

Biotic homogenization: also referred to as the 'anthropogenic blender' (Olden, 2006), the loss of biotic uniqueness, where local community assemblages are becoming more similar to each other on average, and this biotic homogenization (Finderup Nielsen *et al.,* 2019; McKinney & Lockwood, 1999; Yang *et al.,* 2021).

Biotic resistance to invasion: the ability of species in a community to limit the recruitment or invasion of other species (Catford *et al.,* 2009; Levine *et al.,* 2004). It is central to our understanding of how communities at risk of invasion assemble after disturbances, but it has yet to translate into guiding principles for the restoration of invasion-resistant communities (Byun *et al.,* 2013).

Bridging organizations offer a means to improve environmental management outcomes by spanning the science-policy interface to allow for the effective sharing of data, information, and knowledge. Bridging organizations are institutions that use specific mechanisms such as working groups to link and facilitate interactions among individual actors in a management setting." (Kowalski & Jenkins, 2015)

C

Casual species: species that do not have self-sustaining populations and which rely on repeated introductions for their persistence i.e., not yet an established species (Blackburn *et al.,* 2011).

Circular economy: model of production and consumption, which involves sharing, leasing, reusing, repairing, refurbishing and recycling existing materials and products as long as possible. In this way, the life cycle of products is extended (European Parliament, 2015).

Citizen science: diverse range of approaches in which scientific research is conducted, in whole or in part, by volunteers with varying levels of expertise (also known as community science, participatory monitoring, community-based environmental monitoring, crowd science, crowd-sourced science, civic science, or volunteer monitoring). Citizen science often contributes to surveillance of invasive alien species (Gardiner & Roy, 2022; Pocock *et al.,* 2018).

Classical biological control: the intentional introduction of an alien species, usually co-evolved, as a biological control agent for permanent establishment and long-term control (Eilenberg *et al.,* 2001).

Closed water systems: in the context of management of biological invasions, bodies of water that do not directly or indirectly drain with continuous and intensive flow into an ocean or river, recognizing that no natural systems may be entirely closed (e.g., some inland surface waters and water bodies/freshwater).

Collective action: action taken together by a group of people whose goal is to achieve a common objective. It is a term that is used in many areas of the social sciences including psychology, sociology, anthropology, political science and economics (Hardin, 2015).

Colonization pressure: the number of species introduced or released to a single location, some of which will go on to establish a self-sustaining population and some of which will not (Blackburn *et al.,* 2020; Lockwood *et al.,* 2009).

Connected water systems: in the context of management of biological invasions, bodies of water that are directly or indirectly connected to an ocean or a main river (e.g., cryosphere, shelf ecosystems and coastal areas).

Containment: the application of measures in and around an infested area to prevent spread of invasive alien species. Containment may also apply in the context of keeping an invasive alien species out of a defined geographic region within a broader infestation (in pest management this is also termed "area-wide management") (FAO, 2019). Any action taken to delimit the distribution of an invasive alien species through whatever means possible.

Control: direct action(s) taken to reduce or suppress the distribution, abundance, spread and impacts of invasive alien species within a defined geographic area (FAO, 1995) (see management).

Cost-benefit analysis: an analytical tool for judging the economic advantages or disadvantages of an investment decision by assessing its costs and benefits in order to assess the welfare change attributable to it. The analytical framework of CBA refers to a list of underlying concepts which is as follows: opportunity cost, long-term perspective, calculation of economic performance indicators expressed in monetary terms, microeconomic approach, incremental approach (European Commission, 2015).

Cost-effectiveness analysis: an analytical tool to identify the best activity, process, or intervention that justifies/minimizes resource use to achieve a desired result (BetterEvaluation, 2014).

Cryptogenic species: a species, which cannot be reliably demonstrated as being either alien or native (Carlton, 1996).

D

DNA barcoding: a commonly used molecular method (e.g., for detection of species, revealing species interactions and assessment of diversity of community assemblages) that involves the amplification of a short section of DNA from a specific gene or genes. Recent advances have extended the application of this approach from the identification of individual specimens to identification of multiple specimens within mixed samples through DNA metabarcoding (Klink *et al.,* 2022).

E

E-commerce: "online ordering, sale, communication and payment, in particular, business to consumer and consumer to consumer transactions but can also be applicable to business-to-business transactions" (WCO, 2018).

Eco-evolutionary dynamics: reciprocal interactions between ecological and evolutionary processes. Ecological and evolutionary time-scales can be so similar that evolutionary change might be rapid enough to influence ecological dynamics (Brunner *et al.,* 2019; Schoener, 2011).

Ecosystem: a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit (IPBES glossary).

Ecosystem-based management: an environmental management approach that recognizes the full array of interactions within an ecosystem, including humans, rather than considering single issues, species, or ecosystem services in isolation (NOAA, 2020).

Eradication: elimination/extirpation of an invasive alien species from a defined geographic area even in the absence of all preventive measures obviating the necessity for further control measures (Dowdle, 1998). The time period after which an invasive alien species can be considered eradicated depends on the species and location.

Essential Biodiversity Variables: measurement required for study, reporting, and management of biodiversity change (Pereira *et al.,* 2013).

Established alien species: alien species which produce self-sustaining and viable populations for a given period of time, during which climatic extremes typical for the invaded region are experienced, without direct intervention by humans or despite human intervention (Blackburn *et al.,* 2011; Pyšek *et al.,* 2004; Rojas-Sandoval & Acevedo-Rodríguez, 2015).

Externality: an economic concept of uncompensated environmental effects of production and consumption that affect consumer utility and enterprise cost outside the market mechanism (OECD, 2003).

F

Feedback loops: processes that either amplify (positive feedback loop) or diminish (negative feedback loop) the effects of a biological invasion**.** Feedback loops may make the impacts of biological invasions stronger or weaker, starting a chain reaction that repeats again and again. **Negative feedback loop:** A human-natural feedback that continually stabilizes or reduces ongoing or future biological invasions (also known as a 'balancing' feedback loop). **Positive feedback loop:** A human-natural feedback that continually increases ongoing or future biological invasions (also known as 'exacerbating' or 'reinforcing' feedback loops) (Sinclair *et al.,* 2020).

G

Good quality of life: within the context of the IPBES Conceptual Framework – the achievement of a fulfilled human life, a notion which varies strongly across different societies and groups within societies. It is a context-dependent state of individuals and human groups, comprising aspects such as access to food, water, energy and livelihood security, and also health, good social relationships and equity, security, cultural identity, and freedom of choice and action. "Living in harmony with nature", "living-well in balance and harmony with Mother Earth" and "human well-being" are examples of different perspectives on a "Good quality of life" (IPBES glossary).

Governance: the way the rules, norms and actions in a given organization are structured, sustained, and regulated (IPBES glossary).

H

Habitat: "the area, characterized by its abiotic and biotic properties, that is habitable by a particular species" (IUCN Standards and Petitions Committee, 2013).

I

Impacts: changes to nature, nature's contributions to people, and/or the good quality of life (Ricciardi *et al.,* 2013). Impacts can be observed or unobserved. More specifically, impacts to nature (formerly 'ecological impact'), is defined as a measurable change to the properties of an ecosystem (Ricciardi *et al.,* 2013), and implies that all introduced species can have an impact, even when not yet established or widespread, which may vary in magnitude, simply by integration into the ecosystem.

Indigenous and local knowledge systems: social and ecological knowledge practices and beliefs pertaining to the relationship of living beings, including people, with one another and with their environments. Such knowledge can provide information, methods, theory and practice for sustainable ecosystem management (IPBES glossary).

Information and Communication Technology (ICT): a broader term for Information Technology (IT), which refers to all communication technologies, including the internet, wireless networks, cell phones, computers, software, middleware, video-conferencing, social networking, and other media applications and services enabling users to access, retrieve, store, transmit, and manipulate information in a digital form (FAO, 2017b).

Information systems: infrastructures for organising data and information. As examples, the Global Biodiversity Information Facility (GBIF) and Ocean Biogeographic Information System (OBIS) are international on-line infrastructures for organizing data of species presences in space and time. For examples of invasive alien species information systems see Katsanevakis & Roy (2015) and Latombe et al. (2017).

Integrated governance for biological invasions: establishment of relationships between the roles of actors, institutions and instruments, and involving as appropriate all those elements of the socio-ecological system that characterize biological invasion and its management, for the purpose of identifying the strategic interventions needed to improve invasive alien species prevention and control outcomes (definition originated from this assessment, from the thinking on integrated environmental governance).

Integrated pest management: careful consideration of all available pest control techniques and subsequent integration of appropriate measures that discourage the development of pest populations and keep pesticides and other interventions to levels that are economically justified and reduce or minimize risks to human and animal health and the environment. Integrated pest management emphasizes the growth of a healthy crop with the least possible disruption to agro-ecosystems and encourages natural pest control mechanisms (FAO, 2017a). This management method seeks control using the most economical means, and with the least possible hazard to people, property, and the environment (U.S. EPA, 2015b).

Integrated policy for biological invasions: an integrated approach to planning and implementing future options to reduce the spread and limit the impact of biological invasions considers the fact that (1) multiple levels of governance are relevant, (2) diverse actors and decision-makers are involved, (3) the invasion process is multi-staged, and (4) drivers of invasion are multiple and interacting (Herrick, 2019).

Introduction pathway: a suite of processes that result in the introduction of a species from one geographical location to another. It means: 1) geographic routes by which a species is moved outside its natural range (past or present); 2) corridors of introduction (e.g., road, canal, tunnel); and/or 3) human activity that gives rise to an intentional or unintentional introduction. More than one vector (see definition of vector below) within a pathway may be involved in a transfer of species (Pyšek *et al.,* 2011; Genovesi & Shine, 2004).

Invasion cold spot: areas of low alien species richness relative to other regions with similar biogeographic characteristics (O'Donnell *et al.,* 2012). Biodiversity hot spots of diversification and species richness are defined as geographic regions with high diversification rates or high species richness, respectively, while conversely cold spots are geographic regions with low diversification rates or species richness (Melián *et al.,* 2015).

Invasion curve: depiction of the different stages of invasive alien species management from prevention to early detection and eradication, containment and adaptive management (Invasive Species Centre, 2021). The curve shows that eradication of an invasive alien species is less probable and more costly as it spreads over time. Choosing a management action relies on where a species is on the invasion curve.

Invasion debts: the potential increase in biological invasions at a site over a particular time frame in the absence of any interventions (Rouget *et al.,* 2016). It is composed of the number of new species that will be introduced (introduction debt), the number of species that will become invasive (species-based invasion debt), the increase in area affected by invasions (area-based invasion debt), and the increase in the negative impacts caused by introduced species (impact-based invasion debt) (Zengeya & Wilson, 2020).

Invasion hotspot: areas of high alien species richness relative to other regions with similar biogeographic characteristics (O'Donnell *et al.,* 2012). Biodiversity hot spots of diversification and species richness are defined as geographic regions with high diversification rates or high species richness, respectively, while conversely cold spots are geographic regions with low diversification rates or species richness (Melián *et al.,* 2015).

Invasion stages: stages (transport, introduction, establishment, and spread) that a species must pass through on the invasion continuum from native to (invasive) alien species, recognising the need for a species to overcome the barriers (geography, captivity or cultivation, survival, reproduction, dispersal and environmental) that obstruct transition between each stage (Blackburn *et al.,* 2011).

Invasional meltdown: the amplification of impacts of invasive alien species through community-level processes in which there is a cascade of effects, positive feedback loops, arising from the interactions amongst species, in this case alien species, which ultimately affect ecosystem functions (Simberloff, 2006).

Invasive alien species: animals, plants or other organisms introduced directly or indirectly by people into places out of their natural range of distribution, where they have become established and dispersed, and generating an impact on local ecosystems and species (IPBES, 2016); see Chapter 1 for further discussion). Invasive alien species are a subset of established alien species that have negative impacts.

L

Lag phase: the time between when an alien species arrives in a new area and the onset of the phase of rapid, or exponential, increase. Multiple factors are frequently implicated in the persistence or dissolution of the lag phase in biological invasions, including an initial shortage of suitable sites, the absence or shortage of essential mutualists, inadequate genetic diversity, and reduction in competition or predation (due to other alterations in the resident biota) (Zengeya & Wilson, 2020).

Legal personality: any entity that has the ability to conclude and negotiate international agreements in accordance with its external commitments; become a member of international organizations; join international conventions, such as the European Convention on Human Rights, stipulated in Article 6(2) of the Treaty on European Union (EUR-Lex, 2022).

M

Management: for the purpose of the assessment, any action taken to address the threats, risks, distribution, abundance and impacts of an invasive alien species within a defined geographic area (Hulme, 2006; Pyšek *et al.,* 2020). Management includes prevention, preparedness, eradication, containment, and control (Robertson *et al.,* 2020).

Monitoring: for the purpose of this assessment, the continued or regular observation of an ecosystem to detect invasion/reinvasion by invasive alien species and/or their impacts.

N

Native species: taxa that have originated in a given area (their natural range) without human involvement, or that have arrived there without intentional or unintentional intervention of humans, from an area in which they are native (IPBES glossary). This definition excludes products of hybridization involving alien taxa since "human involvement", in this case, includes the introduction of an alien parent (Pyšek *et al.,* 2004).

Nature: in the context of IPBES, refers to the natural world with an emphasis on its living components. Within the context of western science, it includes categories such as biodiversity, ecosystems (both structure and functioning), evolution, the biosphere, humankind's shared evolutionary heritage, and biocultural diversity. Within the context of other knowledge systems, it includes categories such as Mother Earth and systems of life, and it is often viewed as inextricably linked to humans, not as a separate entity (see "Mother Earth") (IPBES glossary).

Nature's contributions to people: all the contributions, both positive and negative, of living nature (i.e., diversity of organisms, ecosystems, and their associated ecological and evolutionary processes) to the quality of life for people. Beneficial contributions from nature include such things as food provision, water purification, flood control, and artistic inspiration, whereas detrimental contributions include disease transmission and predation that damages people or their assets. Many nature's contributions to people may be perceived as benefits or detriments depending on the cultural, temporal or spatial context (IPBES glossary).

Nexus: interlinkages among biodiversity, climate change, adaptation and mitigation including relevant aspects of the energy system, water, food, and health (IPBES, 2021).

O

One Biosecurity: interdisciplinary approach to biosecurity policy and research that builds on the interconnections between human, animal, plant, and environmental health to effectively prevent and mitigate the impacts of invasive alien species. It provides an integrated perspective to address the many biosecurity risks that transcend the traditional boundaries of health, agriculture, and the environment. Individual invasive alien plant and animal species often have multiple impacts across sectors: as hosts of zoonotic parasites, vectors of pathogens, pests of agriculture or forestry, as well as threats to biodiversity and ecosystem function (Hulme, 2020, 2021).

One Health: an integrated, unifying approach that aims to sustainably balance and optimize the health of people, animals, and ecosystems. It recognizes the health of humans, domestic and wild animals, plants, and the wider environment (including ecosystems) are closely linked and interdependent (One Health High-Level Expert Panel (OHHLEP) *et al.,* 2022).

P

Pathway management: any action taken (single or via systems approach) towards a particular anthropogenic invasive alien species arrival pathway (e.g., trade) to prevent or address the threats and risks of an invasive alien species arriving and establishing via that pathway either between or within jurisdictions (Robertson *et al.,* 2020).

Policy: a definite course or method of action selected from among alternatives and in light of given conditions to guide and determine present and future decisions (IPBES, 2019). See also Governance.

Policy cycle: a framework describing the policy process in terms of four linked phases: agenda setting, policy design, policy implementation, and policy review (IPBES glossary).

Policy regime: constructs that depict the mix of institutional mechanisms that make up the governing arrangements addressing a particular problem (Herrick, 2019), noting that for the purpose of this assessment the term "regime" is used for a governance system, affecting more than one country, for a specific issue area, such as invasive alien species (Andonova & Mitchell, 2010).

Polycentric governance: an organizational structure where multiple independent actors mutually order their relationships with one another under a general system of rules (Ostrom, 2010).

Polymerase chain reaction (PCR): sometimes called "molecular photocopying," the polymerase chain reaction (PCR) is a fast and inexpensive technique used to "amplify" - copy - small segments of DNA. Because significant amounts of a sample of DNA are necessary for molecular and genetic analyses, studies of isolated pieces of DNA are nearly impossible without PCR amplification (National Human Genome Research Institute, 2020).

Precautionary approach: where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environment degradation (Principle 15 of the 1992 Rio Declaration, (CBD, 1992).

Preparedness (in the context of invasive alien species management): any policy and/or action undertaken to prepare for the probable arrival of a potential invasive alien species including any preventative or adaptive response activity (Australian Government - Department of Agriculture, Fisheries and Forestry, 2019).

Prevention: for the purpose of this assessment, any policy and/or action/ response undertaken to prevent the arrival and/or introduction of alien and invasive alien species, between and within countries and regions. Prevention is generally far more cost-effective and environmentally beneficial than measures taken following introduction and establishment of an invasive alien species (CBD, 2002).

Propagule pressure (also termed 'introduction effort'): a measure of introduction intensity, including release from captivity or cultivation, comprising both the number of individuals of a species introduced per introduction (propagule size) and the frequency of introductions (Lockwood *et al.,* 2005).

R

Range: "the current limits of distribution of a species, accounting for all known, inferred or projected sites of occurrence" (IUCN, 2016).

Resilience: for the purpose of this assessment, the ability of an ecosystem to adapt, withstand and respond to alien species invasions, recover rapidly from their impacts and continue to develop (U.S. EPA, 2015a).

Restoration: any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state (IPBES glossary). More specifically, in the context of invasive alien species management, it refers to the process of assisting the recovery of a degraded, damaged, or destroyed ecosystem, as a consequence of biological invasions, to reflect values regarded as inherent in the ecosystem and to provide goods and services that people value (adapted from Martin, 2017).

Risk: probability of the occurrence of a particular adverse event at a specific time and the magnitude of the consequent damage caused, depending on various factors such as exposure to the hazard, the frequency of exposure and the severity of any consequent damage done (FAO, 2011b). The term risk is regarded as a product of three factors: Exposure x Likelihood x Consequence (Kinney & Wiruth, 1976). Exposure results from the introductions, establishment and spread of an alien species, whereas Likelihood is the probability of an alien species affecting nature, nature's contributions to people, good quality of life and/or the economy, and Consequence is the magnitude of impacts if an introduction event occurs (D'hondt *et al.,* 2015).

S

Safe trade: export of products that are free from invasive alien species (Burgiel *et al.,* 2006).

Sentinel sites or locations: selected locations with heightened levels of detection and effective reporting through concentration of activities on subpopulations to enhance detection and improve cost-effectiveness of invasive alien species surveillance efforts (Keeling *et al.,* 2017).

Site-based management: programmes that aim to manage the impacts of invasive alien species within a site/area through both implementation of control measures and where necessary restoration (sometimes referred to as asset protection) e.g., within high value protected sites/areas.

Species-led management: invasive alien species management (in all contexts) focused on reducing the threats and impacts of specific or multiple invasive alien species.

Surveillance: actions, including extended programme of surveys and general surveillance (capturing unstructured and untargeted surveillance data and information from a wide range of sources), undertaken in order to directly or indirectly detect the presence of one or many invasive alien species over time (CEPM, 1996; Clift, 2008; CPM, 2015).

T

Tragedy of the commons: a situation in which individuals with access to a public resource (also called a common) act in their own interest and, in doing so, ultimately deplete the resource (Spiliakos, 2019).

Transformative change: a fundamental, system-wide reorganization across technological, economic, and social factors making sustainability the norm (Díaz *et al.,* 2019).

Transformative governance: the set of formal and informal (public and private) rules, rulemaking systems and actor networks at all levels of human society that enable transformative change (Visseren-Hamakers *et al.,* 2021).

Trend: temporal trends are directional long-term changes (i.e., decades to centuries) in numbers of species, populations or individuals introduced, or the spatial extent of colonization (Buckland *et al.,* 2017). In this assessment report, trends are presented as indicators of species numbers (species richness) and rates of accumulation of species (e.g., first records of a species in a given location) over time.

V

Value chains (that link production systems, markets and consumers): a contact network, which provides opportunities for the transmission of contagious diseases within and between sectors. It follows that these chains (networks) can be understood and taken into account in planning risk management strategies for disease prevention and control" especially in relation with "risky parts of the value chain" (FAO, 2011a).

Vector: Any living or non-living carrier that transports living organisms intentionally or unintentionally (ICES, 2005).

W

Widespread species: species that are able to maintain viable populations across a range of environments leading to a large range size. Widespread species are likely to experience a large range of ecological and climatic conditions within their range. A large niche width – based on the current distribution of a species – seems to be a general pattern in widespread species (Gaston, 2003; Vincent *et al.,* 2020).

Willingness to pay: the stated price that an individual would accept to pay for avoiding the loss or the diminution of an environmental service (United Nations, 2003).

GLOSSARY REFERENCES

Andonova, L. B., & Mitchell, R. B. (2010). The Rescaling of Global Environmental Politics. *Annual Review of Environment and Resources*, *35*(1), 255–282. https://doi.org/10.1146/annurev-environ-100809-125346

Australian Government - Department of Agriculture, Fisheries and Forestry. (2019, December 31). *Biosecurity Preparedness Plans* [Department of Agriculture, Fisheries and Forestry]. https://www.agriculture.gov.au/fees/biosecurity-preparedness-plans

BetterEvaluation. (2014). *Cost Effectiveness Analysis*. BetterEvaluation.Org. https://www.betterevaluation.org/en/evaluation-options/CostEffectivenessAnalysis

Blackburn, T. M., Cassey, P., & Duncan, R. P. (2020). Colonization pressure: A second null model for invasion biology. *Biological Invasions*, *22*(4), 1221–1233. https://doi.org/10.1007/s10530-019-02183-7

Blackburn, T. M., Pyšek, P., Bacher, S., Carlton, J. T., Duncan, R. P., Jarošík, V., Wilson, J. R. U., & Richardson, D. M. (2011). A proposed unified framework for biological invasions. *Trends in Ecology & Evolution*, *26*(7), 333–339. https://doi.org/10.1016/j.tree.2011.03.023

Brunner, F. S., Deere, J. A., Egas, M., Eizaguirre, C., & Raeymaekers, J. A. M. (2019). The diversity of eco-evolutionary dynamics: Comparing the feedbacks between ecology and evolution across scales. *Functional Ecology*, *33*(1), 7–12. https://doi.org/10.1111/1365-2435.13268

Buckland, S. T., Yuan, Y., & Marcon, E. (2017). Measuring temporal trends in biodiversity. *AStA Advances in Statistical Analysis*, *101*(4), 461–474. https://doi.org/10.1007/s10182-017-0308-1

Burgiel, S., Foote, G., Orellana, M., & Perrault, A. (2006). *Invasive Alien Species and Trade: Integrating Prevention Measures and International Trade Rules* (p. 54). Center for International Environmental Law (CIEL) and Defenders of Wildlife. https://cleantrade.typepad.com/clean_trade/files/invasives_trade_paper_0106.pdf

Byun, C., De Blois, S., & Brisson, J. (2013). Plant functional group identity and diversity determine biotic resistance to invasion by an exotic grass. *Journal of Ecology*, *101*(1), 128–139. https://doi.org/10.1111/1365-2745.12016

Carlton, J. T. (1996). Biological Invasions and Cryptogenic Species. *Ecology*, *77*(6), 1653–1655. https://doi.org/10.2307/2265767

Catford, J. A., Jansson, R., & Nilsson, C. (2009). Reducing redundancy in invasion ecology by integrating hypotheses into a single theoretical framework. *Diversity and Distributions*, *15*(1), 22–40. https://doi.org/10.1111/j.1472-4642.2008.00521.x

CBD. (1992). *Preamble to the Convention on Biological Diversity.*

CBD. (2002, April 7). Alien species that threaten ecosystems, habitats or species. *COP 6 Decision VI/23*. COP 6, The Hague, Netherlands. https://www.cbd.int/decision/cop/?id=7197

CCBA. (2008). *Climate, Community & Biodiversity Project Design Standards Second Edition*. CCBA. www.climate-standards.org.

CEPM. (1996). *Report of the third meeting of the committee of experts on phytosanitary measures*. Food and Agriculture Organization of The United Nations.

Clift, K. H. (2008). What does the future hold for surveillance? In K. J. Froud, A. I. Popay, & S. M. Zydenbos (Eds.), *Surveillance for biosecurity: Pre-border to pest management*. The New Zealand Plant Protection Society. https://nzpps.org/_oldsite/books/2008_Surveillance/Surveillance.pdf

CPM. (2015, December). *CPM 10—2015*. International Plant Protection Convention - Commission on Phytosanitary Measures. https://www.ippc.int/en/cpm-sessions/cpm-10-2015/

D'hondt, B., Vanderhoeven, S., Roelandt, S., Mayer, F., Versteirt, V., Adriaens, T., Ducheyne, E., San Martin, G., Grégoire, J.-C., Stiers, I., Quoilin, S., Cigar, J., Heughebaert, A., & Branquart, E. (2015). Harmonia+ and Pandora+: Risk screening tools for potentially invasive plants, animals and their pathogens. *Biological Invasions*, *17*(6), 1869–1883. https://doi.org/10.1007/s10530-015-0843-1

Díaz, S., Settele, J., Brondízio, E. S., Ngo, H. T., Agard, J., Arneth, A., Balvanera, P., Brauman, K. A., Butchart, S. H. M., Chan, K. M. A., Garibaldi, L. A., Ichii, K., Liu, J., Subramanian, S. M., Midgley, G. F., Miloslavich, P., Molnár, Z., Obura, D., Pfaff, A., … Zayas, C. N. (2019). Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science*, *366*(6471). https://doi.org/10.1126/science.aax3100

Dowdle, W. R. (1998). The principles of disease elimination and eradication. *Bulletin of the World Health Organization*, *76*(Suppl 2), 22–25.

Eilenberg, J., Hajek, A., & Lomer, C. (2001). Suggestions for unifying the terminology in biological control. *BioControl*, *46*(4), 387–400. https://doi.org/10.1023/A:1014193329979

EUR-Lex. (2022, March 23). *Glossary of summaries. Legal personality of the Union*. EUR-Lex. Access to European Union Law. https://eur-lex.europa.eu/EN/legal-content/glossary/legal-personality-of-the-union.html

European Commission (Ed.). (2015). *Guide to cost-benefit analysis of investment projects: Economic appraisal tool for cohesion policy 2014-2020*. European Union.

European Parliament. (2015, February 12). *Circular economy: Definition, importance and benefits*. News. European Parliament. https://www.europarl.europa.eu/news/en/headlines/economy/20151201STO05603/circular-economy-definition-importance-and-benefits

FAO. (1995). *ISPM 5. Glossary of phytosanitary terms*.

FAO. (2007). *FAO Biosecurity Toolkit. Part 1 Biosecurity principles and components* (p. 20). Food and Agriculture Organization of the United Nations (FAO). https://www.fao.org/3/a1140e/a1140e01.pdf

FAO. (2011a). *A value chain approach to animal diseases risk management – Technical foundations and practical framework for field application* (N. Taylor & J. Rushton, Eds.). Food and Agriculture Organization of the United Nations.

FAO. (2011b). *Biosafety resource book—Module C: risk analysis: Vol. C*. Food and Agriculture Organization of the United Nations. https://www.fao.org/3/i1905e/i1905e02.pdf

FAO. (2017a). *Integrated Pest Management of major pest and diseases in Eastern Europe and Caucasus*. FAO. https://www.fao.org/publications/card/en/c/ee1c6ac7-1186-458f-a95f-0e6d69dea858/

FAO. (2019). *Glossary of phytosanitary terms. International Standard for Phytosanitary Measures No. 5* (p. 35). Published by FAO on behalf of the Secretariat of the International Plant Protection Convention (IPPC).

FAO. (2017b). *Information and Communication Technologies (ICT) | AIMS*. Food and Agriculture Organization of the United Nations (FAO). http://aims.fao.org/information-and-communication-technologies-ict

Finderup Nielsen, T., Sand-Jensen, K., Dornelas, M., & Bruun, H. H. (2019). More is less: Net gain in species richness, but biotic homogenization over 140 years. *Ecology Letters*, *22*(10), 1650–1657. https://doi.org/10.1111/ele.13361

Folke, C., Hahn, T., Olsson, P., & Norberg, J. (2005). Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources*, *30*(1), 441–473. https://doi.org/10.1146/annurev.energy.30.050504.144511

Gardiner, M. M., & Roy, H. E. (2022). The Role of Community Science in Entomology. *Annual Review of Entomology*, *67*(1), 437–456. https://doi.org/10.1146/annurev-ento-072121-075258

Gaston, K. J. (2003). *The structure and dynamics of geographic ranges*. Oxford University Press on Demand.

Gavin, M. C., McCarter, J., Mead, A., Berkes, F., Stepp, J. R., Peterson, D., & Tang, R. (2015). Defining biocultural approaches to conservation. *Trends in Ecology and Evolution*, *30*(3), 140–145. https://doi.org/10.1016/j.tree.2014.12.005

Genovesi, P., & Shine, C. (2004). *European strategy on invasive alien species: Convention on the Conservation of European Wildlife and Habitats (Bern Convention)* (Nature and Environment No. 137). Council of Europe. https://www.cbd.int/doc/external/cop-09/bern-01-en.pdf

Hardin, R. (2015). *Collective Action* (0 ed.). RFF Press. https://doi.org/10.4324/9781315044330

Herrick, C. N. (2019). A review of the U.S. invasive species policy mix: Questioning the prospect of an integrated regime. *Environmental Policy and Governance*, *29*(4), 262–278. https://doi.org/10.1002/eet.1852

Hulme, P. E. (2006). Beyond control: Wider implications for the management of biological invasions. *Journal of Applied Ecology*, *43*(5), 835–847. https://doi.org/10.1111/j.1365-2664.2006.01227.x

Hulme, P. E. (2020). One Biosecurity: A unified concept to integrate human, animal, plant, and environmental health. *Emerging Topics in Life Sciences*, *4*(5), 539–549. https://doi.org/10.1042/ETLS20200067

Hulme, P. E. (2021). Advancing one biosecurity to address the pandemic risks of biological invasions. *BioScience*, *71*(7), 708–721. https://doi.org/10.1093/biosci/biab019

ICES. (2005). *ICES Code of Practice on the Introductions and Transfers of Marine Organisms 2005* (p. 30). https://www.nobanis.org/globalassets/ices-code-of-practice.pdf

Invasive Species Centre. (2021). *Investing in Prevention*. Invasive Species Centre. https://www.invasivespeciescentre.ca/

IPBES. (2016). *Scoping report for a thematic assessment of invasive alien species* (p. 8). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPBES. (2019). *Methodological guidance for assessing policy instruments and facilitating the use of policy support tools and methodologies through IPBES assessments (module D of the Guide on the production of assessments)*.

IPBES. (2021). *Annex I to decision IPBES-8/1*.

IUCN. (2016). *A global standard for the identification of Key Biodiversity Areas: Version 1.0*. IUCN. https://portals.iucn.org/library/node/46259

IUCN Standards and Petitions Committee. (2013). *Guidelines for Using the IUCN Red List Categories and Criteria.* (Version 10.1). Prepared by the Standards and Petitions Committee. Downloadable from http://www.iucnredlist.org/documents/RedListGuidelines.pdf.

Katsanevakis, S., & Roy, H. (2015). Alien species related information systems and information management. *Management of Biological Invasions*, *6*(2), 115–117. https://doi.org/10.3391/mbi.2015.6.2.01

Keeling, M. J., Datta, S., Franklin, D. N., Flatman, I., Wattam, A., Brown, M., & Budge, G. E. (2017). Efficient use of sentinel sites: Detection of invasive honeybee pests and diseases in the UK. *Journal of The Royal Society Interface*, *14*(129), 20160908. https://doi.org/10.1098/rsif.2016.0908

Kinney, G. F., & Wiruth, A. D. (1976). *Practical Risk Analysis for Safety Management*. Naval Weapons Center, China Lake, CA,. https://www.researchgate.net/profile/Guibert-Crevecoeur/post/What-is-the-unification-theory-of-safety-science/attachment/59d6252c79197b80779837dc/AS%3A316678640996352%40145251347 9333/download/Basic_Kinney_article.pdf

Klink, R. van, August, T., Bas, Y., Bodesheim, P., Bonn, A., Fossøy, F., Høye, T. T., Jongejans, E., Menz, M. H. M., Miraldo, A., Roslin, T., Roy, H. E., Ruczyński, I., Schigel, D., Schäffler, L., Sheard, J. K., Svenningsen, C., Tschan, G. F., Wäldchen, J., … Bowler, D. E. (2022). Emerging technologies revolutionise insect ecology and monitoring. *Trends in Ecology & Evolution*, *0*(0). https://doi.org/10.1016/j.tree.2022.06.001

Kowalski, A., & Jenkins, L. (2015). The role of bridging organizations in environmental management: Examining social networks in working groups. *Ecology and Society*, *20*(2). https://doi.org/10.5751/ES-07541-200216

Latombe, G., Pyšek, P., Jeschke, J. M., Blackburn, T. M., Bacher, S., Capinha, C., Costello, M. J., Fernández, M., Gregory, R. D., Hobern, D., Hui, C., Jetz, W., Kumschick, S., McGrannachan, C., Pergl, J., Roy, H. E., Scalera, R., Squires, Z. E., Wilson, J. R. U., … McGeoch, M. A. (2017). A vision for global monitoring of biological invasions. *Biological Conservation*, *213*, 295–308. https://doi.org/10.1016/j.biocon.2016.06.013

Levine, J. M., Adler, P. B., & Yelenik, S. G. (2004). A meta-analysis of biotic resistance to exotic plant invasions. *Ecology Letters*, *7*(10), 975–989. https://doi.org/10.1111/j.1461-0248.2004.00657.x

Lockwood, J. L., Cassey, P., & Blackburn, T. M. (2005). The role of propagule pressure in explaining species invasions. *Trends in Ecology & Evolution*, *20*(5), 223–228. https://doi.org/10.1016/j.tree.2005.02.004

Lockwood, J. L., Cassey, P., & Blackburn, T. M. (2009). The more you introduce the more you get: The role of colonization pressure and propagule pressure in invasion ecology. *Diversity and Distributions*, *15*(5), 904–910. https://doi.org/10.1111/j.1472-4642.2009.00594.x

LPP and LIFE Network, (LPP), L. for P. P., Development, E. L., Network, L., & LPP and LIFE Network. (2010). *Biocultural Community Protocols for Livestock Keepers*. Lokhit Pashu-Palak Sansthan. http://www.fao.org/3/a-ax407e.pdf

Lyver, P. O., Timoti, P., Davis, T., & Tylianakis, J. M. (2019). Biocultural Hysteresis Inhibits Adaptation to Environmental Change. *Trends in Ecology & Evolution*, *34*(9), 771–780. https://doi.org/10.1016/j.tree.2019.04.002

Martin, D. M. (2017). Ecological restoration should be redefined for the twenty-first century. *Restoration Ecology*, *25*(5), 668–673. https://doi.org/10.1111/rec.12554

McKinney, M. L., & Lockwood, J. L. (1999). Biotic homogenization: A few winners replacing many losers in the next mass extinction. *Trends in Ecology & Evolution*, *14*(11), 450–453. https://doi.org/10.1016/S0169-5347(99)01679-1

Melián, C. J., Seehausen, O., Eguíluz, V. M., Fortuna, M. A., & Deiner, K. (2015). Diversification and biodiversity dynamics of hot and cold spots. *Ecography*, *38*(4), 393–401. https://doi.org/10.1111/ecog.01162

National Human Genome Research Institute. (2020). *Polymerase Chain Reaction (PCR) Fact Sheet*. Genome.Gov. https://www.genome.gov/about-genomics/fact-sheets/Polymerase-Chain-Reaction-Fact-Sheet

NOAA. (2020, August 18). *National Oceanic and Atmospheric Administration (NOAA) Ecosystem-Based Management*. https://ecosystems.noaa.gov/

O'Donnell, J., Gallagher, R. V., Wilson, P. D., Downey, P. O., Hughes, L., & Leishman, M. R. (2012). Invasion hotspots for non-native plants in Australia under current and future climates. *Global Change Biology*, *18*(2), 617–629. https://doi.org/10.1111/j.1365-2486.2011.02537.x

OECD. (2003). *OECD Glossary of Statistical Terms*. OECD Glossary of Statistical Terms. https://stats.oecd.org/glossary/index.htm

Olden, J. D. (2006). Biotic homogenization: A new research agenda for conservation biogeography. *Journal of Biogeography*, *33*(12), 2027–2039. https://doi.org/10.1111/j.1365-2699.2006.01572.x

One Health High-Level Expert Panel (OHHLEP), Adisasmito, W. B., Almuhairi, S., Behravesh, C. B., Bilivogui, P., Bukachi, S. A., Casas, N., Becerra, N. C., Charron, D. F., Chaudhary, A., Zanella, J. R. C., Cunningham, A. A., Dar, O., Debnath, N., Dungu, B., Farag, E., Gao, G. F., Hayman, D. T. S., Khaitsa, M., … Zhou, L. (2022). One Health: A new definition for a sustainable and healthy future. *PLOS Pathogens*, *18*(6), e1010537. https://doi.org/10.1371/journal.ppat.1010537

Ostrom, E. (2010). Polycentric systems for coping with collective action and global environmental change. *Global Environmental Change*, *20*(4), 550–557. https://doi.org/10.1016/j.gloenvcha.2010.07.004

Pereira, H. M., Ferrier, S., Walters, M., Geller, G. N., Jongman, R. H. G., Scholes, R. J., Bruford, M. W., Brummitt, N., Butchart, S. H. M., Cardoso, A. C., Coops, N. C., Dulloo, E., Faith, D. P., Freyhof, J., Gregory, R. D., Heip, C., Hoft, R., Hurtt, G., Jetz, W., … Wegmann, M. (2013). Essential Biodiversity Variables. *Science*, *339*(6117), 277–278. https://doi.org/10.1126/science.1229931

Pocock, M. J. O., Chandler, M., Bonney, R., Thornhill, I., Albin, A., August, T., Bachman, S., Brown, P. M. J., Cunha, D. G. F., Grez, A., Jackson, C., Peters, M., Rabarijaon, N. R., Roy, H. E., Zaviezo, T., & Danielsen, F. (2018). A Vision for Global Biodiversity Monitoring With Citizen Science. In *Advances in Ecological Research* (Vol. 59, pp. 169–223). Elsevier. https://doi.org/10.1016/bs.aecr.2018.06.003

Pyšek, P., Hulme, P. E., Simberloff, D., Bacher, S., Blackburn, T. M., Carlton, J. T., Dawson, W., Essl, F., Foxcroft, L. C., Genovesi, P., Jeschke, J. M., Kühn, I., Liebhold, A. M., Mandrak, N. E., Meyerson, L. A., Pauchard, A., Pergl, J., Roy, H. E., Seebens, H., … Richardson, D. M. (2020). Scientists' warning on invasive alien species. *Biological Reviews*, *95*(6), 1511–1534. https://doi.org/10.1111/brv.12627

Pyšek, P., Jarošík, V., & Pergl, J. (2011). Alien plants introduced by different pathways differ in invasion success: Unintentional introductions as a threat to natural areas. *PLoS ONE*, *6*(9), e24890. https://doi.org/10.1371/journal.pone.0024890

Pyšek, P., Richardson, D. M., Rejmánek, M., Webster, G. L., Williamson, M., & Kirschner, J. (2004). Alien plants in checklists and floras: Towards better communication between taxonomists and ecologists. *TAXON*, *53*(1), 131–143. https://doi.org/10.2307/4135498

Ricciardi, A., Hoopes, M. F., Marchetti, M. P., & Lockwood, J. L. (2013). *Progress toward understanding the ecological impacts of nonnative species. 83*(3), 263–282. https://doi.org/10.1890/13-0183.1

Richardson, D. M., Pyšek, P., & Carlton, J. T. (2010). A Compendium of Essential Concepts and Terminology in Invasion Ecology. In D. M. Richardson (Ed.), *Fifty Years of Invasion Ecology* (pp. 409–420). Wiley-Blackwell. https://doi.org/10.1002/9781444329988.ch30

Robertson, P. A., Mill, A., Novoa, A., Jeschke, J. M., Essl, F., Gallardo, B., Geist, J., Jarić, I., Lambin, X., Musseau, C., Pergl, J., Pyšek, P., Rabitsch, W., von Schmalensee, M., Shirley, M., Strayer, D. L., Stefansson, R. A., Smith, K., & Booy, O. (2020). A proposed unified framework to describe the management of biological invasions. *Biological Invasions*, *22*(9), 2633–2645. https://doi.org/10.1007/s10530-020-02298-2

Rojas-Sandoval, J., & Acevedo-Rodríguez, P. (2015). Naturalization and invasion of alien plants in Puerto Rico and the Virgin Islands. *Biological Invasions*, *17*(1), 149–163. https://doi.org/10.1007/s10530-014-0712-3

Rouget, M., Robertson, M. P., Wilson, J. R. U., Hui, C., Essl, F., Renteria, J. L., & Richardson, D. M. (2016). Invasion debt—Quantifying future biological invasions. *Diversity and Distributions*, *22*(4), 445–456. https://doi.org/10.1111/ddi.12408

Schoener, T. W. (2011). The newest synthesis: Understanding the interplay of evolutionary and ecological dynamics. *Science*, *331*(6016), 426–429. https://doi.org/10.1126/science.1193954

Simberloff, D. (2006). Invasional meltdown 6 years later: Important phenomenon, unfortunate metaphor, or both? *Ecology Letters*, *9*(8), 912–919. https://doi.org/10.1111/j.1461-0248.2006.00939.x

Sinclair, J. S., Brown, J. A., & Lockwood, J. L. (2020). Reciprocal human-natural system feedback loops within the invasion process. *NeoBiota*, *62*, 489–508. https://doi.org/10.3897/neobiota.62.52664

Spiliakos, A. (2019, February 6). *Tragedy of the Commons: Examples & Solutions | HBS Online*. Business Insights Blog. https://online.hbs.edu/blog/post/tragedy-of-the-commons-impact-on-sustainability-issues

United Nations, E. C. (2003). *Integrated Environmental and Economic Accounting 2003*. United Nations New York.

U.S. EPA. (2015a). *Environmental Resilience: Exploring Scientific Concepts for Strengthening Community Resilience to Disasters.* (EPA/600/R-15/163). U.S. Environmental Protection Agency.

U.S. EPA. (2015b, September 28). *Integrated Pest Management (IPM) Principles* [Overviews and Factsheets]. https://www.epa.gov/safepestcontrol/integrated-pest-management-ipm-principles

Vincent, H., Bornand, C. N., Kempel, A., & Fischer, M. (2020). Rare species perform worse than widespread species under changed climate. *Biological Conservation*, *246*, 108586. https://doi.org/10.1016/j.biocon.2020.108586

Visseren-Hamakers, I. J., Razzaque, J., McElwee, P., Turnhout, E., Kelemen, E., Rusch, G. M., Fernández-Llamazares, Á., Chan, I., Lim, M., Islar, M., Gautam, A. P., Williams, M., Mungatana, E., Karim, M. S., Muradian, R., Gerber, L. R., Lui, G., Liu, J., Spangenberg, J. H., & Zaleski, D. (2021). Transformative governance of biodiversity: Insights for sustainable development. *Current Opinion in Environmental Sustainability*, *53*, 20–28. https://doi.org/10.1016/j.cosust.2021.06.002

WCO. (2018). *Cross-Border E-Commerce Framework of Standards*. World Customs Organization.

Yang, Q., Weigelt, P., Fristoe, T. S., Zhang, Z., Kreft, H., Stein, A., Seebens, H., Dawson, W., Essl, F., König, C., Lenzner, B., Pergl, J., Pouteau, R., Pyšek, P., Winter, M., Ebel, A. L., Fuentes, N., Giehl, E. L. H., Kartesz, J., … van Kleunen, M. (2021). The global loss of floristic uniqueness. *Nature Communications*, *12*(1), 7290. https://doi.org/10.1038/s41467-021-27603-y

Zélé, F., Magalhães, S., Kéfi, S., & Duncan, A. B. (2018). Ecology and evolution of facilitation among symbionts. *Nature Communications*, *9*(1), Article 1. https://doi.org/10.1038/s41467-018-06779-w

Zengeya, T. A., & Wilson, J. R. (2020). *The status of biological invasions and their management in South Africa in 2019* (Final Report). Zenodo. https://doi.org/10.5281/ZENODO.3947613

ANNEX II
Acronyms

ASEAN	Association of Southeast Asian Nations
AU$	Australian dollar
BWM Convention	International Convention for the Control and Management of Ships' Ballast Water and Sediments
CABI	Centre for Agriculture and Biosciences International
CARE	Collective Benefit, Authority to Control, Responsibility and Ethics
CBD	Convention on Biological Diversity
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CO_2	Carbon dioxide
COP	Conference of the Parties
COVID-19	Coronavirus disease 2019
CRISPR	Clustered Regularly Interspaced Short Palindromic Repeats
DIISE	Database of Island Invasive Species Eradications
DNA	Deoxyribonucleic Acid
DPSR	Driver- Pressure- State- Response
dsRNA	double-stranded Ribonucleic Acid
EICAT	Environmental Impact Classification for Alien Taxa
€	Euro
FAIR	Findable, Accessible, Interoperable and Reusable
FAO	Food and Agriculture Organization (of the United Nations)
GDP	Gross Domestic Product
GEF	Global Environment Facility
GEO BON	Group on Earth Observations Biodiversity Observation Network
GISD	Global Invasive Species Database
GloNAF	Global Naturalized Alien Flora
GRIIS	Global Register of Introduced and Invasive Species
IMO	International Maritime Organization
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IPPC	International Plant Protection Convention
ISPM	International Standards for Phytosanitary Measures
ISSG	Invasive Species Specialist Group
IUCN	International Union for Conservation of Nature
MERCOSUR	Mercado Común del Sur (Southern Common Market)
NBSAPs	National Biodiversity Strategies and Action Plans
NZ$	New Zealand dollars
PCR	Polymerase Chain Reaction
RHDV	Rabbit Haemorrhagic Disease Virus
RNAi	Ribonucleic Acid Interference
SARS-CoV-2	Severe Acute Respiratory Syndrome Coronavirus 2
SCOPE	Scientific Committee on Problems of the Environment
SDGs	Sustainable Development Goals
SEICAT	Socio-Economic Impact Classification for Alien Taxa
SIDS	Small Island Developing States
SPS Agreement	Agreement on the Application of Sanitary and Phytosanitary Measures
SSC	Species Survival Commission
UNESCO	United Nations Educational, Scientific, and Cultural Organization
UNFCCC	United Nations Framework Convention on Climate Change
US$	United States dollars
WCO	World Customs Organization
WHO	World Health Organization
WOAH	World Organisation for Animal Health
WTO	World Trade Organization

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