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.

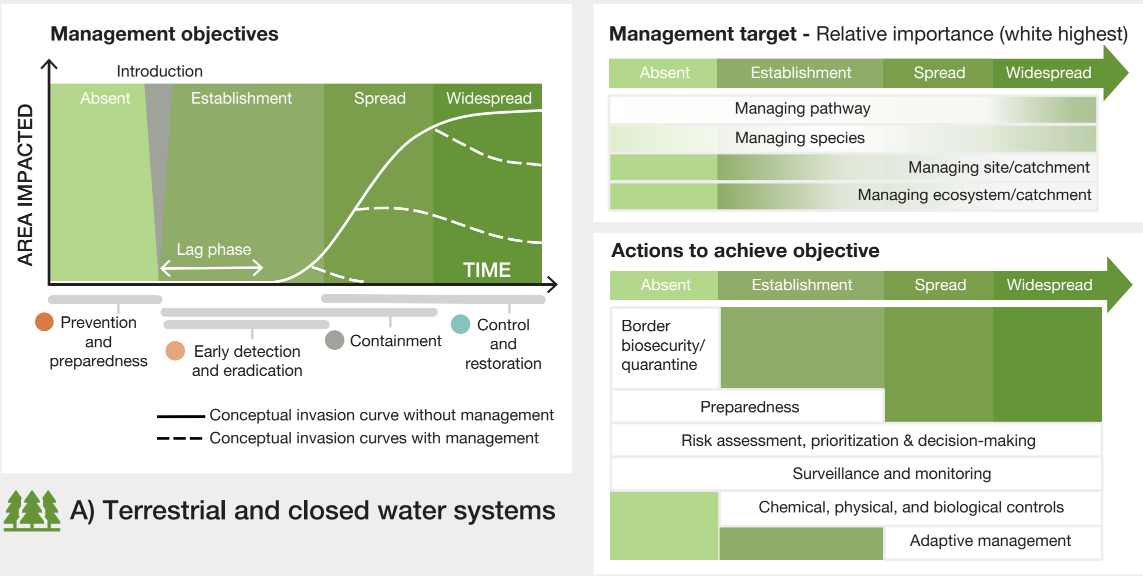
Section 5.5 assesses examples of successful and unsuccessful management approaches and examines how evidence-based decision-making and modern tools and technologies have brought successful management outcomes. It also provides evidence on management costs.

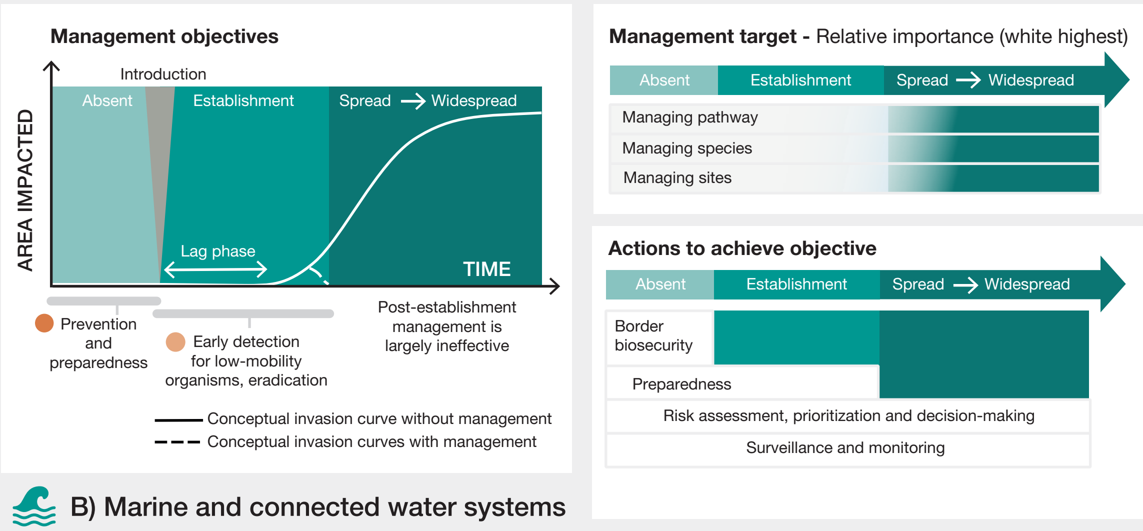
Section 5.6 summarizes the challenges in achieving effective management of biological invasions. It emphasizes context dependency and perspectives of various stakeholders and Indigenous Peoples and local communities on invasive alien species and related social conflict and discusses the knowledge gaps, 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

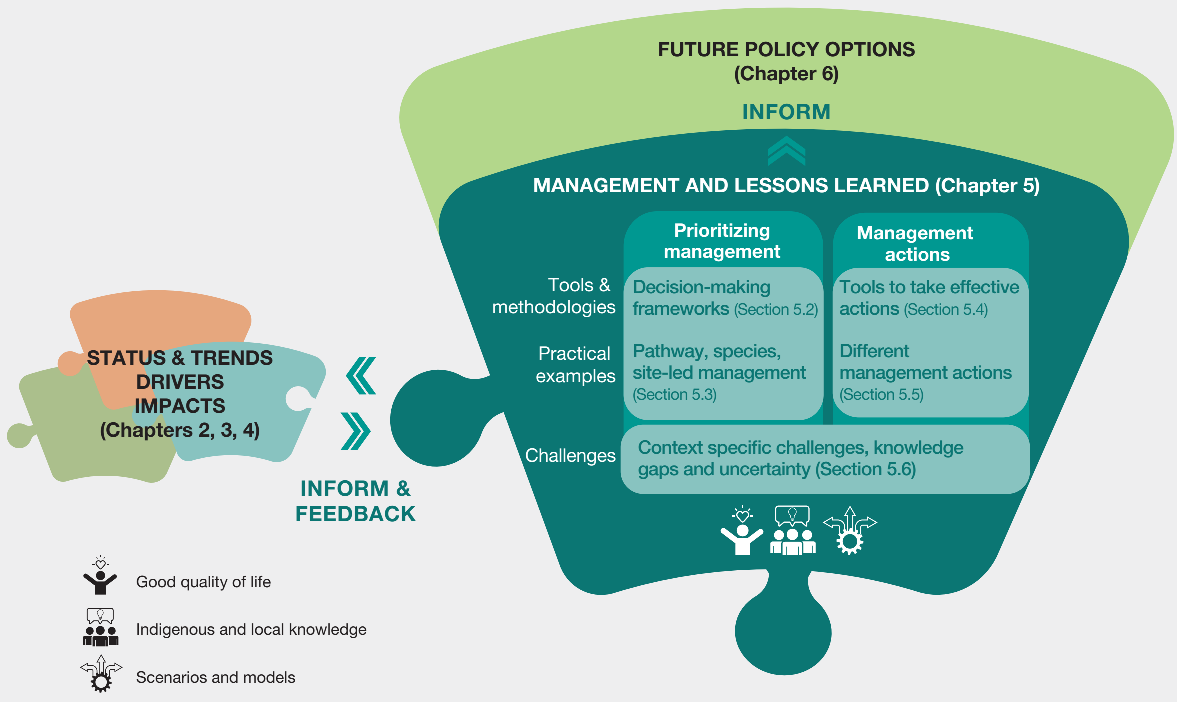


Figure 5 2 **Content of sections in this chapter and their linkages with the other chapters of the assessment.**

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

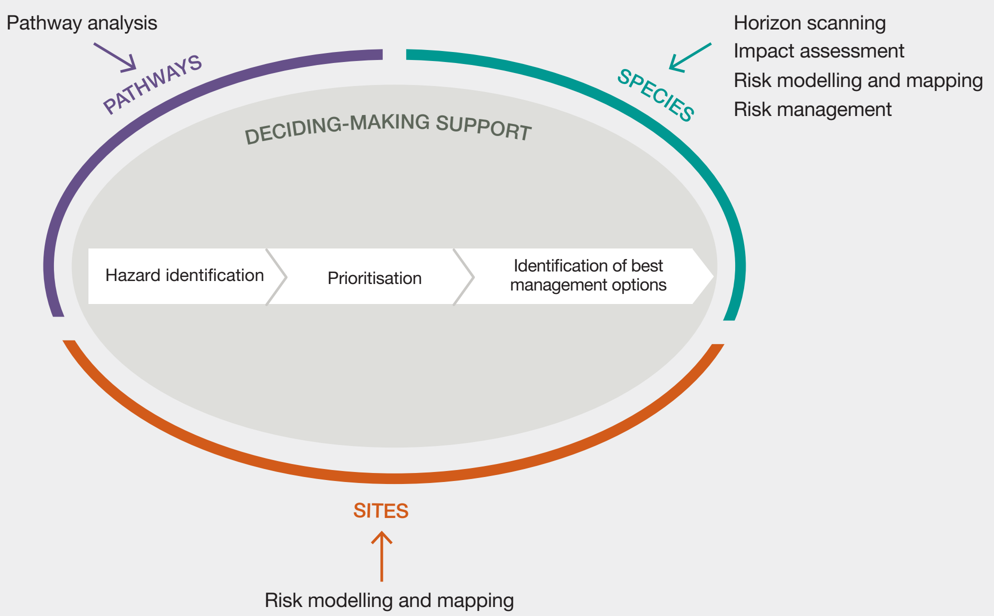


Figure 5 3 **The three main approaches to support decision-making actions on management of biological invasions and examples of contributing tools.**

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

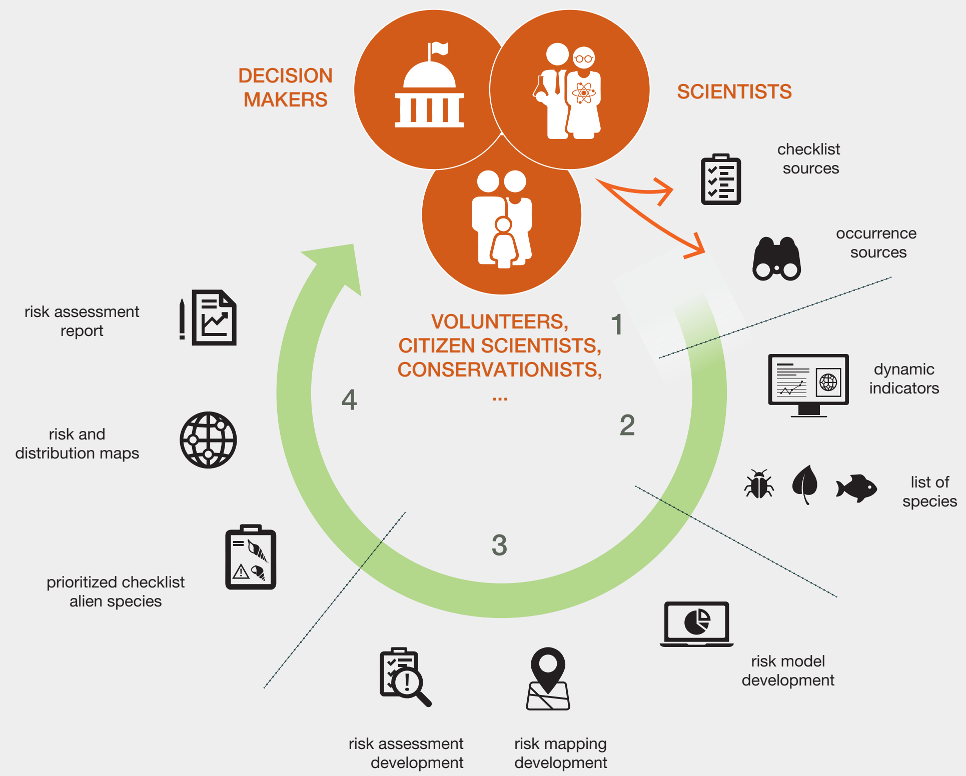


Figure 5 5 **The TrIAScycle – an example of knowledge flow allowing for transparency and promoting trust and confidence between stakeholders and decision makers.**

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

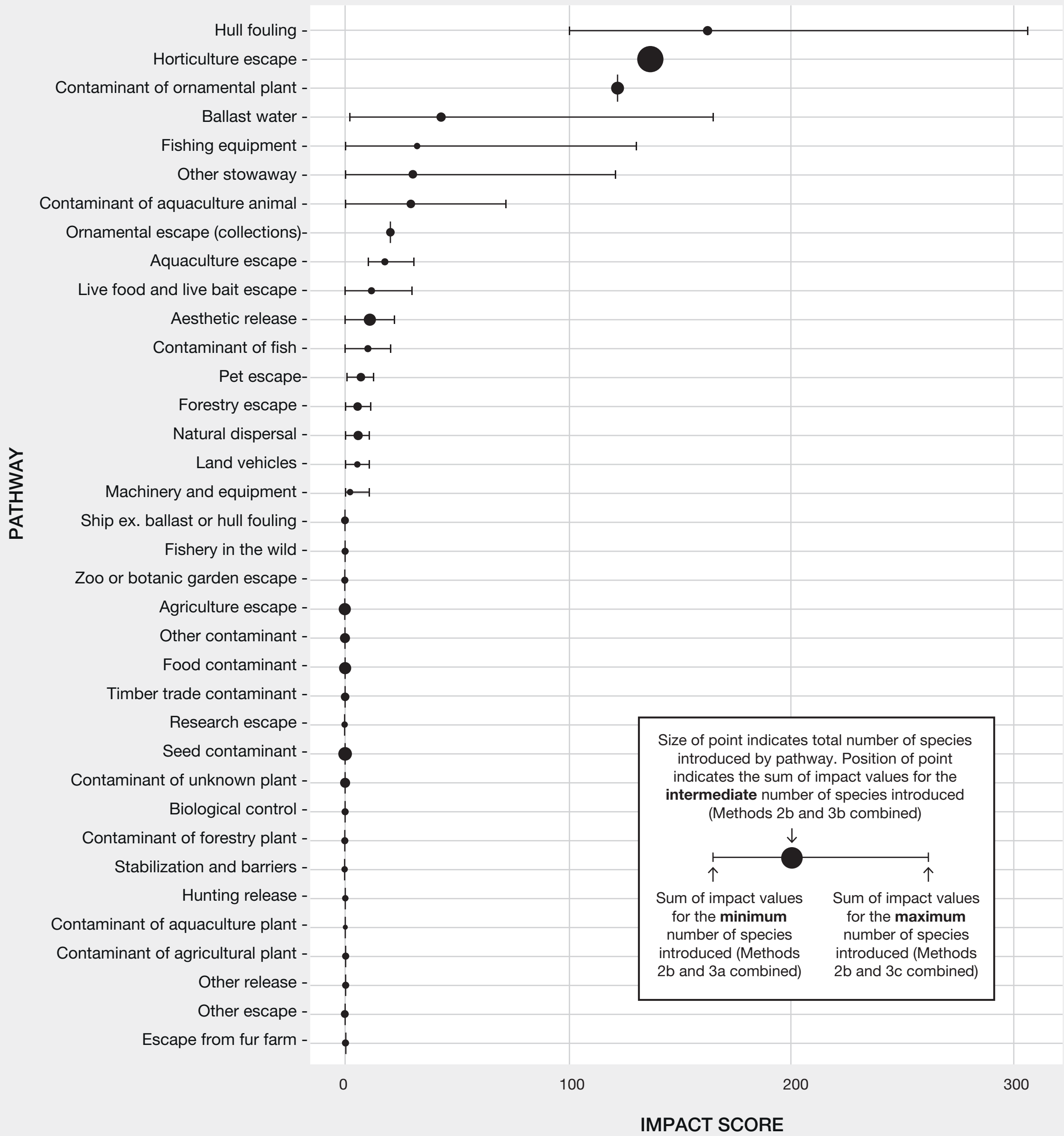


Figure 5 7 **Example of a pathway prioritization with a case study from Great Britain (United Kingdom).**

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

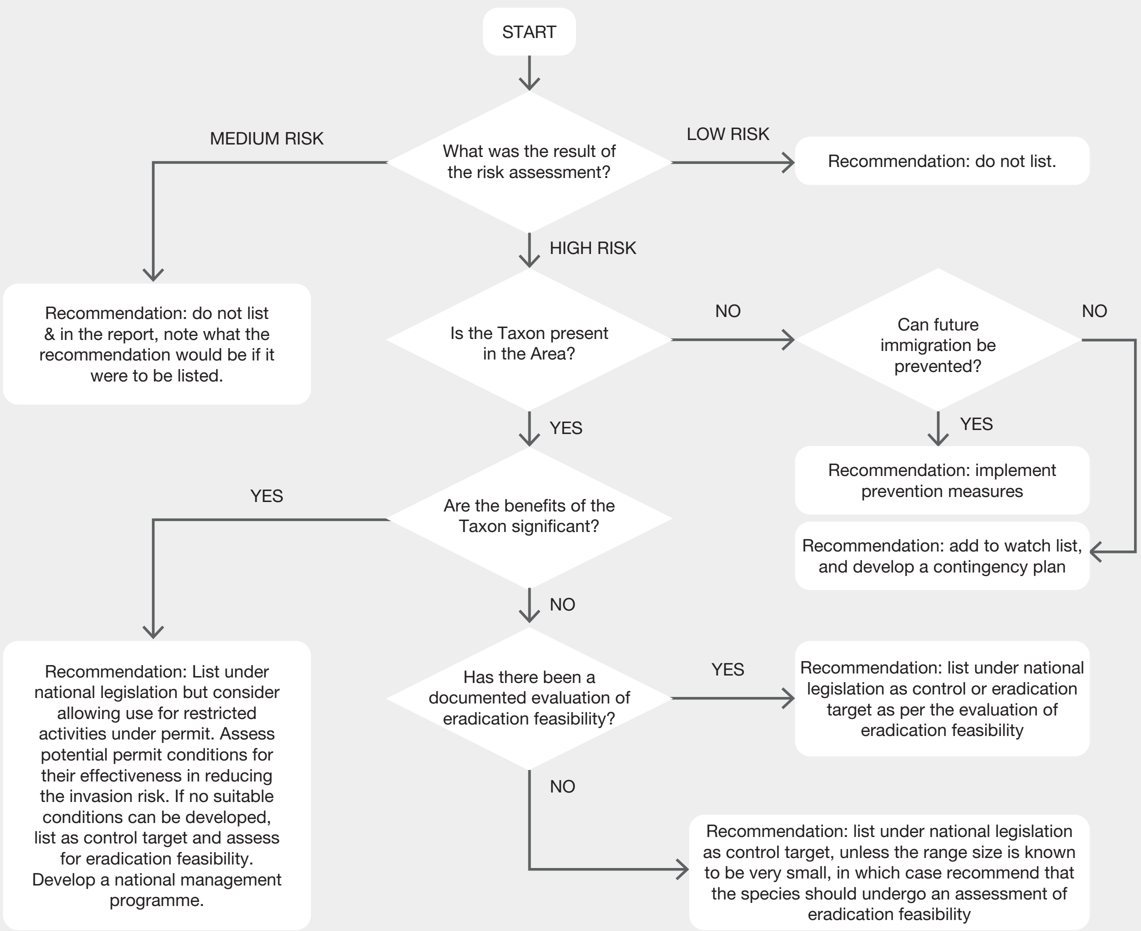


Figure 5 8 **Risk Analysis for Alien Taxa (RAAT) framework developed in South Africa as a standardized and transparent approach to prioritizing and regulating alien species based on evidence.**

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.

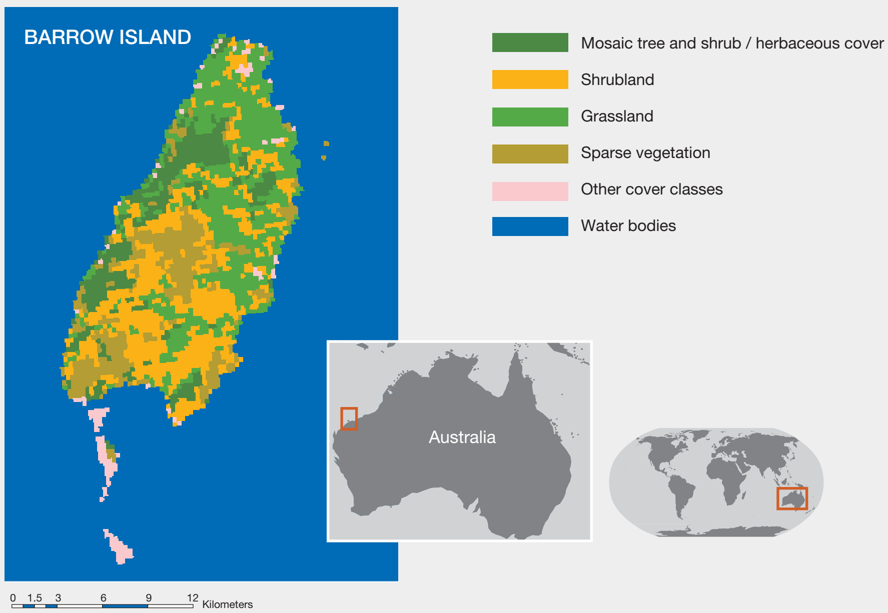


Figure 5.9 **Map of Barrow Island in north-western Australia which is dominated by grasslands and has an area of 202 km².**

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**).

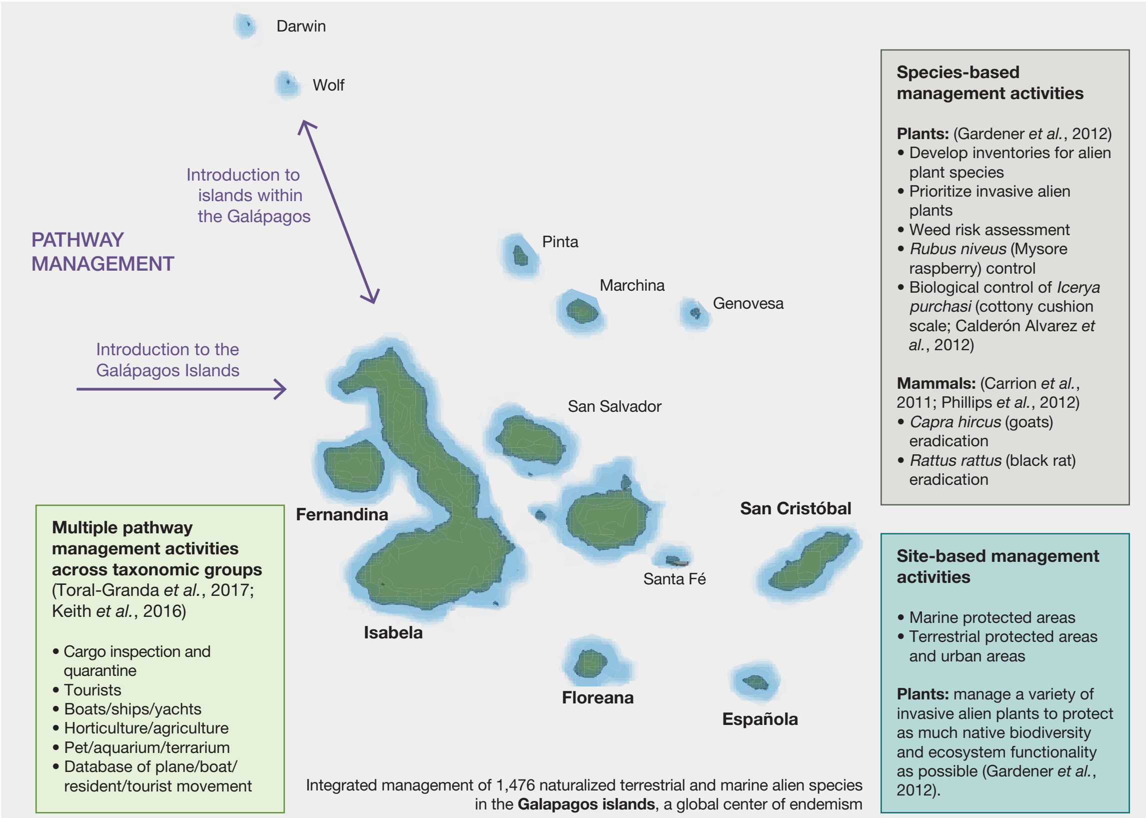


Figure 5 12 **Map of the Galápagos Islands showing examples of pathway, species-based and site-based management activities.**

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) –

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)**.

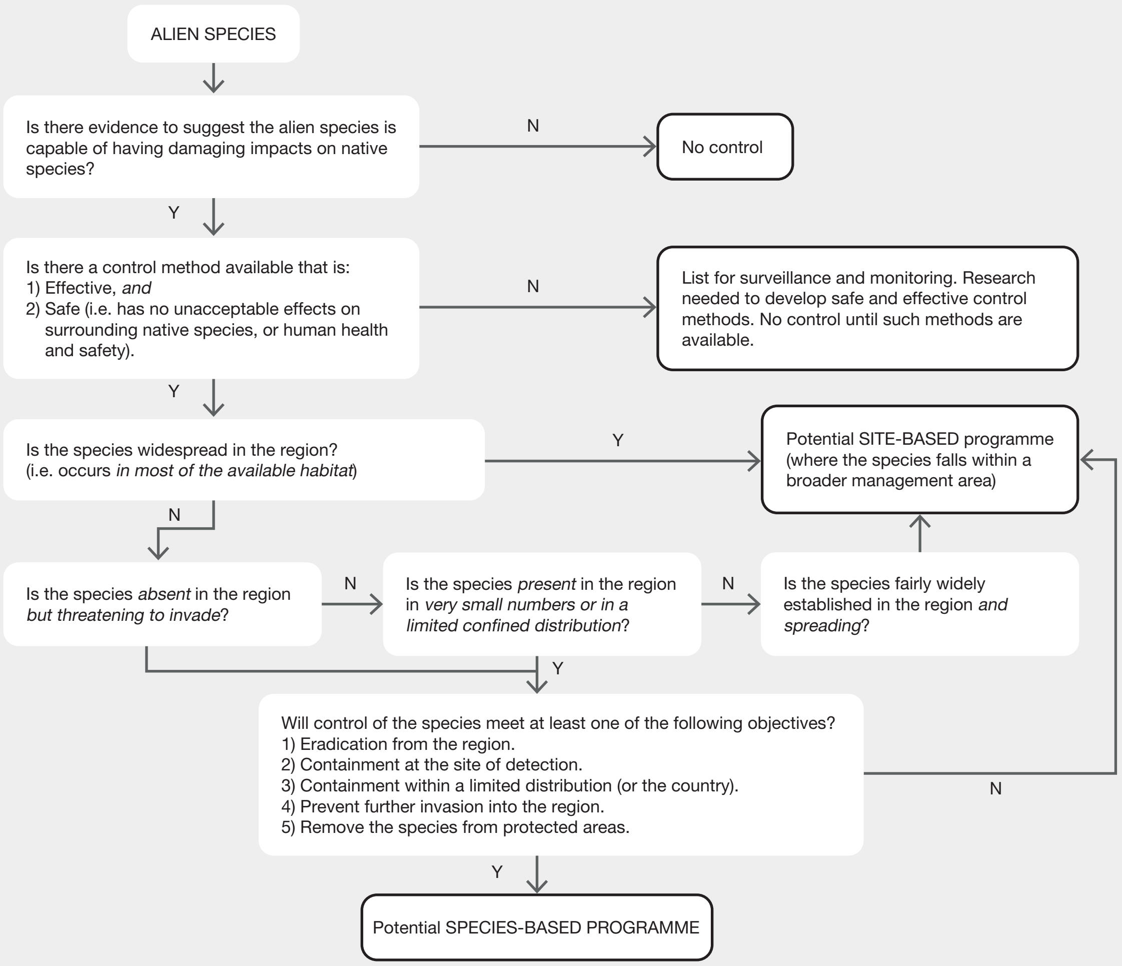


Figure 5 14 **Decision tree for choosing between site-based and species-based management of invasive alien plants.**

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

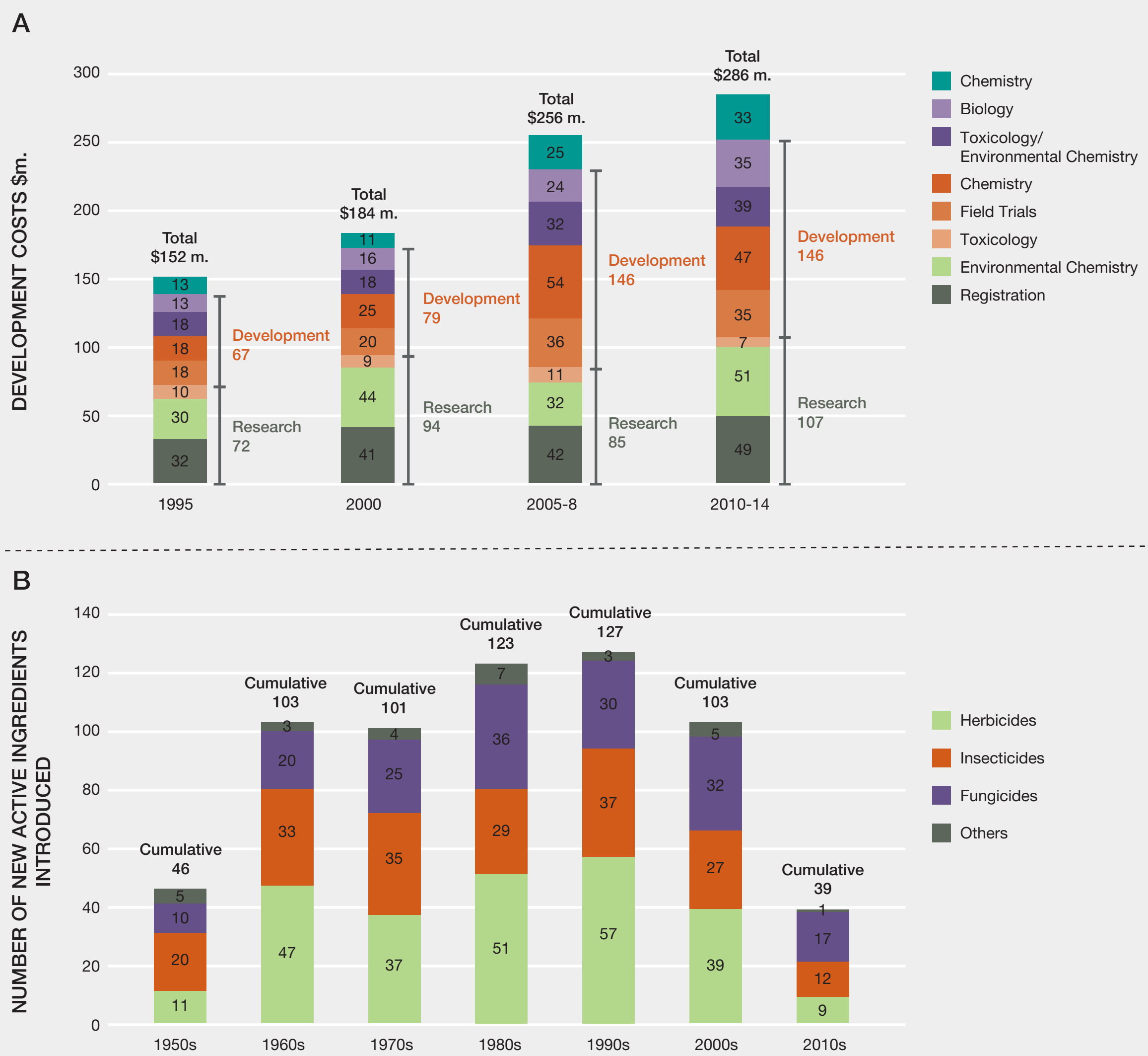


Figure 5 16 **Evolution of new crop protection product.**

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

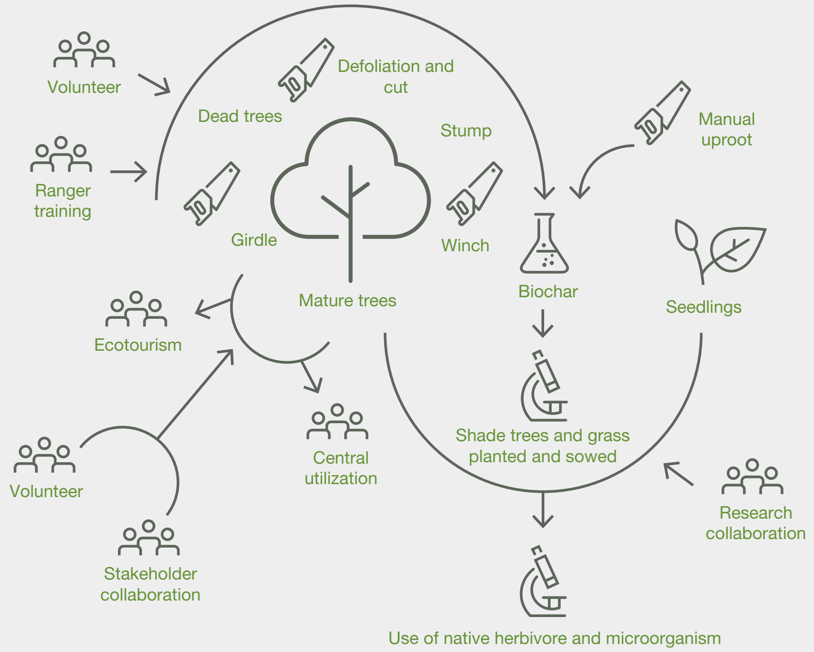


Figure 5 19 **Plan for integration of stakeholder engagement and management strategies for *Vachellia nilotica* (gum arabic tree) management in Baluran National Park, Indonesia.**

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

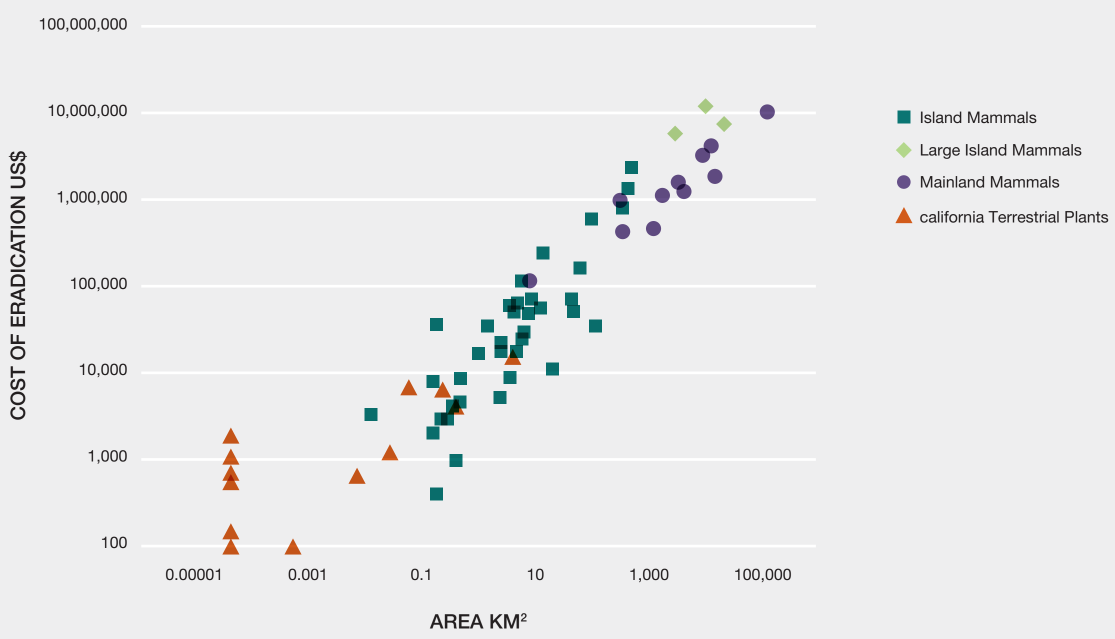


Figure 5 23 **Successful eradication programmes; the relationship between area (km^2, x axis) of successful eradication campaigns and eradication programme cost (y axis).**

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).

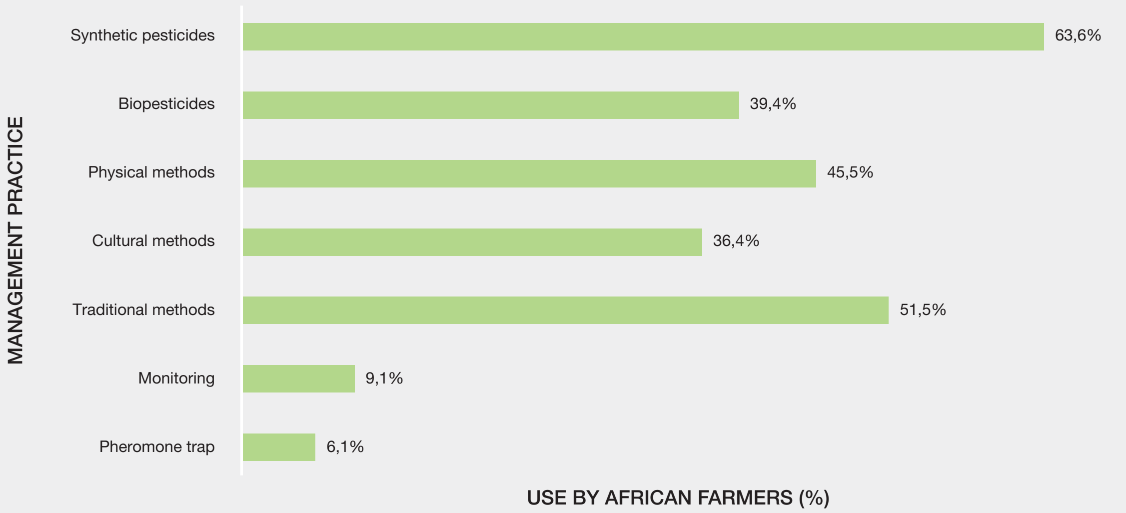


Figure 5 24 **Fall armyworm management adopted by African farmers uses combinations of chemical, physical, cultural or traditional methods.**

(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.

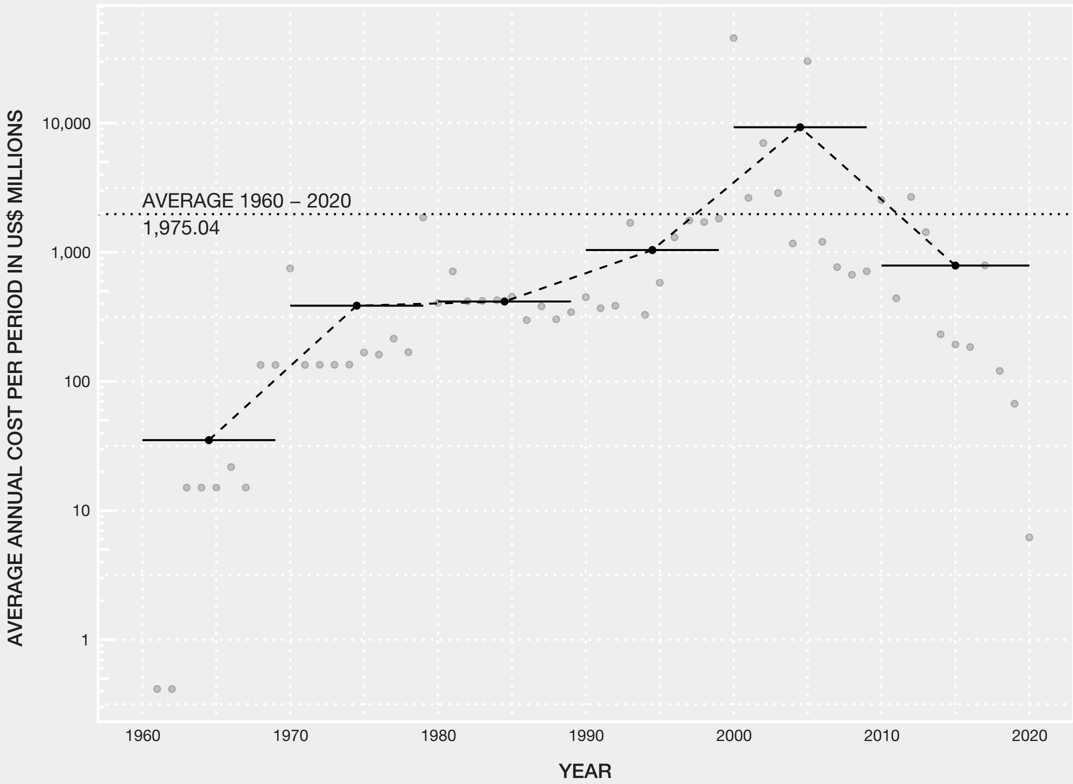


Figure 5 25 **Annual cost reported globally with the management of biological invasions between 1960 and 2020 (2017 monetary values).**

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).

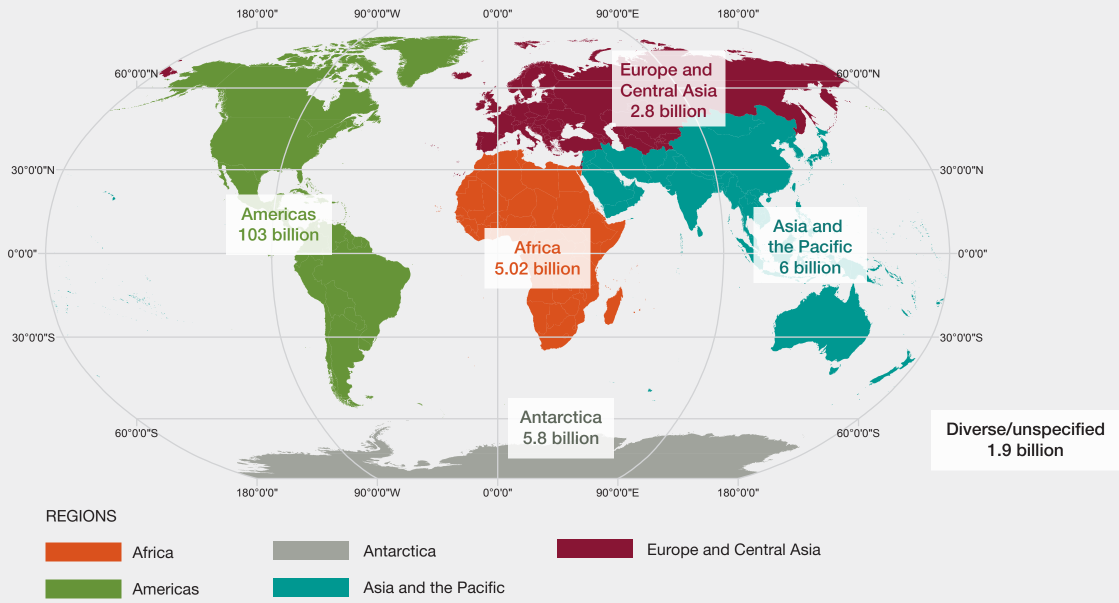


Figure 5 26 **Total reported economic cost of management of biological invasions actions across all IPBES regions (in USD) between 1960 and 2020 (2017 monetary values).**

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

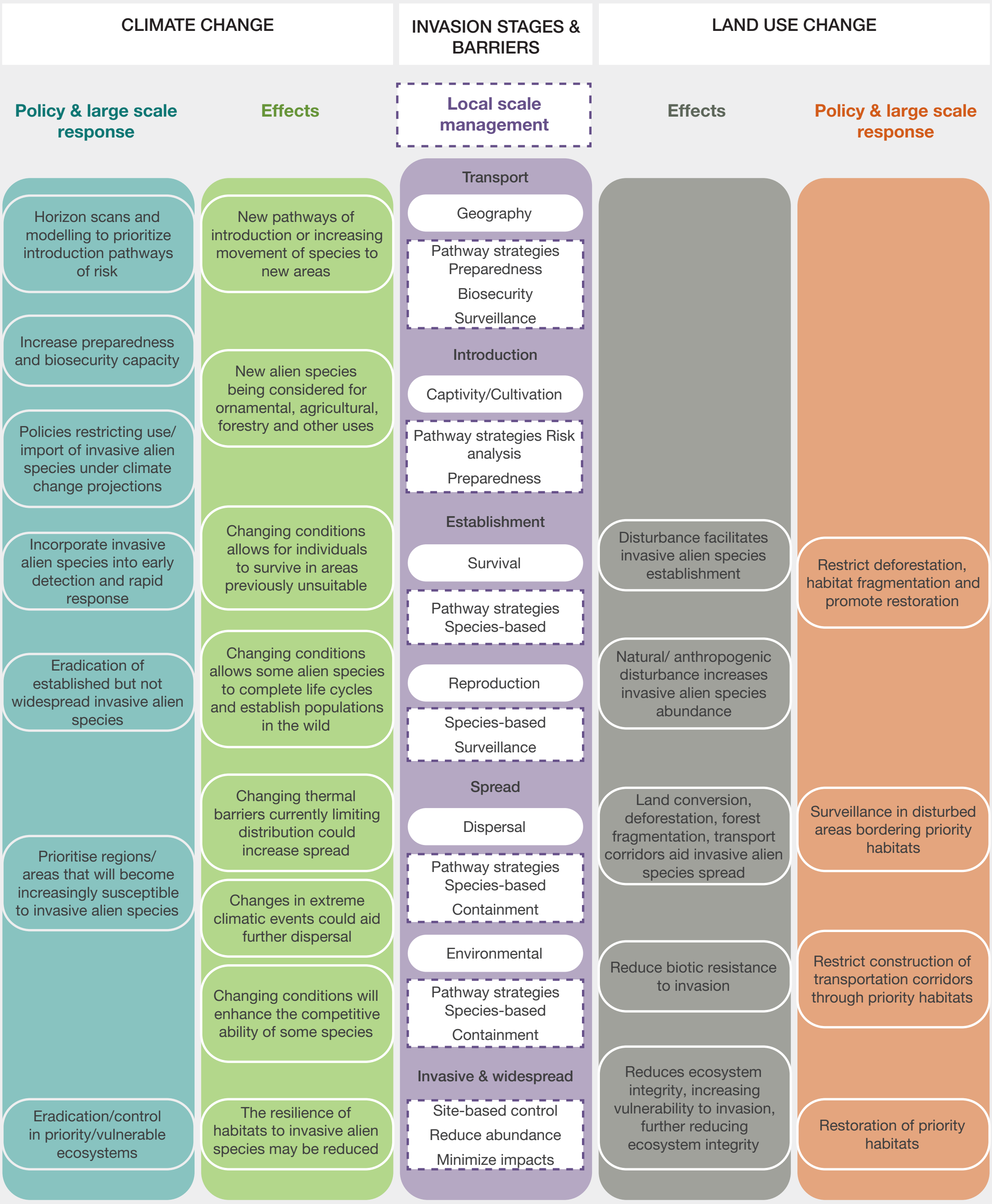


Figure 5 27 **Climate change and habitat transformation interactions at various stages of the biological invasion process, and potential management responses.**

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
Pathway Management	**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
Pathway Management	**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**
Pathway Management	**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.

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