# Ocean monetary values to support adaptive MSP

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# Contents

# 1 Introduction

Marine spatial planning (MSP) is a planning process that uses Ecosystem Based Management (EBM) principles and focuses on the spatially explicit nature of many ocean activities and resources (TEEB 2012, p23). EBM differs from traditional approaches focused on single sectors, activities or species, by taking account of interactions, synergies and cumulative effects. MSP needs to take account of the services provided and potentially provided from different areas, the activities involved in accessing them, and the resulting cumulative effects on marine ecosystems. The planning approach should be ecosystem based and spatially explicit, and should consider human benefits and impacts, address cumulative impacts, and take account of future activities and changes, with the aim of ensuring that the collective pressure of activities remains compatible with a healthy and sustainable marine environment (Nordic Council of Ministers 2017).

Services from the deep sea are in increasing demand, and pressure to utilize more fully deep-sea products such as seafood, energy resources and minerals are on the rise (Thurber, Sweetman et al. 2014). The deep North Atlantic Ocean is now known to harbour ecosystems that support a biologically rich variety of life that perform key functions within global biogeochemical cycles (Armstrong et al, 2019a). The deep-sea ecosystems, including cold water corals, sponges, seamounts and hydrothermal vents, also provide many other ecosystem goods and services, which contribute to maritime economic activities that underpin the socio-economic well-being of Atlantic nations and their citizens (Galparsoro et al, 2014; Armstrong et al, 2019a). These services include nutrient cycling, waste absorption and detoxification, fisheries, bioprospecting and a number of cultural services related to education and science, aesthetic and inspirational contributions (Armstrong et al, 2012).

However, marine ecosystems and resources are subject to significant pressures. Human activities, but also climate change effects, and natural hazards and dynamics such as erosion and accretion, can have severe impacts on marine ecosystems, leading to deterioration of environmental status, loss of biodiversity and degradation of ecosystem services (COM 2014). These pressures and impacts in turn have potentially significant consequences for marine economic development and growth. The dual recognition that human pressures directly impact on ecosystem services and that ecosystem services directly benefit human well-being has led to increasing efforts to integrate ecosystem services in policy and management (Galparsoro, Borja et al. 2014).

Achieving sustainable exploitation of marine resources in the deep sea is particularly challenging, due to the huge uncertainty around the many risks posed by human activities on these remote and relatively poorly understood ecosystems (Armstrong et al, 2019a), for which management regimes are often poorly defined, in particular in the areas beyond national jurisdiction (ABNJ). There are often difficult trade-offs to make between different possible services and the immediate and longer-term impacts of marine activities (Armstrong et al 2019a).

It is essential to consider the various pressures and their impacts in the establishment of marine spatial plans (COM 2014). So in order to evaluate the effectiveness and sustainability of a plan for simultaneously benefiting from and conserving marine resources, a range of ecological, socio-economic and institutional indicators need to be developed and monitored (Douvere and Ehler, 2011). These indicators must include the identification of services, their values and conflict areas, and their incorporation as important inputs to policy making, and in particular marine spatial planning (Armstrong et al, 2014)

To date, however, there is a lack of environmental baselines and assessments in relation to human interactions with the deep sea (Armstrong et al, 2019a). Consequently MSP is not well developed for the deep sea, and most existing MSP focuses on coastal waters or shelf areas. With growing anthropogenic pressures in deep-sea environments, developing sustainable plans is a priority. Better knowledge of the values provided by habitat-based sea-floor ecosystem services could help to justify further policy action, development of Marine Protected Areas, conservation, and resource use.[1] This information could also help design responses to global change that will inevitably impact on deep-sea

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[1] Global Ocean Commission, 2014

ecosystems and biodiversity, and the services they provide.[2] The ATLAS project has started to put in place the information required for economic baselines in the North Atlantic, considering areas both inside and outside EEZs. The research includes:

- Identification of ecosystem goods and services (Deliverable 5.1)
- Assessment of risks to ecosystem services from diverse human drivers (Deliverable 5.2):
- Ecosystem goods and services and environmental risk assessment (Deliverable 6.2):
- Original stated-preference valuation surveys for two ATLAS case-study areas (Deliverable 5.4)
- A Q study of decision-maker and stakeholder views on the legitimacy, validity and acceptability of monetary valuation methods and the use of values in decision support (Deliverable 5.3)

Together, the results of this work can be considered as a first step towards establishing an economic baseline for adaptive MSP in the deep North Atlantic Ocean.

# 2 Policy context

The European MSP Directive (COM 2014) defines MSP as a process through which the relevant authorities analyse and organise human activities in marine areas to achieve ecological, economic and social objectives. Furthermore, the Directive calls for an adaptive management approach to ensure refinement and further development as experience and knowledge increase. An adaptive approach is particularly relevant with regard the deep sea where complete knowledge, data and information are not available. It is also relevant as knowledge grows on the pressures occurring on ecosystem services from human activities (Armstrong et al, 2019a; ATLAS D5.2).

Thus MSP can be thought of as a "public process of analyzing and allocating the spatial and temporal distribution of human activities in marine areas to achieve ecological, economic, and social objectives" (Ehler & Douvere 2009, p24). This involves considering a wide range of different systems, features, activities and services with complex interactions, and reaching difficult decisions about priorities where there are trade-offs or competing objectives, such as protecting biodiverse deep-sea habitats while at the same time safeguarding marine commercial interests and human livelihoods.

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[2] ATLAS Policy Brief "Recognising connectivity and climate change impacts: essential elements for an effective North-Atlantic MPA network"

However, there are challenges to implementing MSP: governance structures are often complex, conflicting and incomplete, and our understanding of ecosystem functions and processes, how they sustain benefits and how they are impacted by human activities is often limited. Assessing ecological processes and resources in terms of the ecosystem goods and services they provide helps to translate highly complex systems into a series of measurements which can be more readily understood by a wide range of stakeholders. But quantitative assessment of cumulative impacts and multiple stressors on ecosystems and services remain the exception rather than the norm.[3] Furthermore, though the identification of trade-offs between ecosystem services may in some cases be apparent, the actual assessment for the balancing of these trade-offs is challenging, when the services have incommensurable measures. In addition, the trade-offs may not only be between services, but also between different stakeholders invested in the same or different services. Single services may be allocated over time and space, further complicating distribution. Broadening out the stakeholder portfolio is a central part of the work in WP5, including the general public, a group that perceives themselves as clear stakeholders, also in relation to the deep sea (Armstrong et al 2019b).

One tool that can help in seeking appropriate balances is the monetary valuation of non-marketed ecosystem goods and services. The ubiquitous nature of trade-offs and conflicting objectives means that identification of services, their values and conflict areas are important for policy making, and for MSP in particular, with new opportunities to balance uses and protection of marine ecosystems in support of the implementation of ecosystem-based management, in line with evolving policies. This report focuses on the role of monetary valuation in supporting MSP.

## 2.1 Policy drivers for valuation

Several international and national initiatives promote greater use of monetary valuation for improving decision making. Notably, these include the Aichi targets[4] under the Convention on Biological Diversity (CBD), in particular:

- **Aichi Target 2:** By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.

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[3] https://roa.midatlanticocean.org/ocean-ecosystem-and-resources/characterizing-the-mid-atlantic-ocean-ecosystem/ecosystem-services/

[4] https://www.cbd.int/sp/targets/

- **Aichi Target 19:** By 2020, knowledge, the science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.

At the European level, the EU Biodiversity Strategy[5] (Target 2, Action 5) called for the mapping and assessment of ecosystems and their services by all Member States (MS). This also included the assessment of the economic value of such services where possible by 2020. A common system of typologies of ecosystems and services for mapping and inclusion in natural capital accounting was developed to be applied by the EU and MS in order to ensure consistent approaches (Bouwma et al. 2018).

At the same time, MS are developing their National Marine Spatial Plans as required under the EU Maritime Spatial Planning Directive (EC Directive 2014/89/EU). The availability of information on ecosystem service delivery at different spatial scales is essential for this process, to ensure that marine economic activities are conducted in a way that sustains the long-term capacity of the oceans to deliver ecosystem services. Recent changes to the Marine Strategy Framework Directive (MSFD) (EC Directive 2008/56/EC) have made MS reporting requirements more explicit regarding information on the human pressures on marine ecosystems and taking into account recent scientific progress (Cavallo et al 2019). The amendments have also introduced a risk-based approach to the reporting requirements for the MSFD through the use of threshold values based on the precautionary principle, helping MS to assess areas at risk that may need further action in achieving Good Environmental Status. The MSFD also requires the availability of ecosystem services valuation for the assessment of the environmental status and to define the measures that ensure sustainable human activities at sea (Cardoso et al 2010).

As part of this process, the ecosystem services provided by benthic habitats in the European North Atlantic Ocean have been assessed and mapped in the context of the "Mapping and Assessment of Ecosystems and their Services" (MAES[6]) programme, the European Biodiversity Strategy and the implementation of the MSFD (see Galparsoro et al 2014). However, the mapping of marine ecosystems services and associated benefits stills lags behind the terrestrial counterparts, and this is

[5] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52011DC0244

[6] https://biodiversity.europa.eu/maes

even more so for deep-water ecosystem services. Most of the recommended indicators[7] for ecosystem services delivered by marine ecosystems fall under 'marine inlets and transitional waters' followed by 'coastal waters', with few listed for 'open ocean' or 'shelf waters'. The EEA has reported (EEA 2016) that for marine ecosystems, the information base "is poor and fragmented, so that assessment at the European level remains challenging", while within Member States' reporting there was "some lack of clarity in the ecosystem typology, in particular with regards to marine ecosystems". These issues need to be resolved before better integration of ecosystem service valuation in policy can be achieved.

## 2.2 Atlantic deep sea management

Sustainable development of the ocean is a central policy objective in Europe through the Blue Growth Strategy, and globally through parties to the CBD (Armstrong et al, 2019a). The EU Blue Growth Strategy, formulated in 2012, requires maritime spatial planning to ensure efficient and sustainable management of activities at sea. "Blue growth" is the concept of encouraging development of marine economic activities, in a sustainable manner, such that the long-term ability of the marine environment to continue to provide ecosystem services is not compromised. The EU Blue Growth strategy seeks to harness the potential of Europe's oceans, seas and coasts for growth and jobs. The aim was to drive forward the EU's Integrated Maritime Policy (IMP) by promoting the EU's blue economy (Mulazzani and Malorgio 2017). Hence, the strategy aims to contribute to the EU's competitiveness, resource efficiency, job creation and new sources of growth whilst safeguarding biodiversity and protecting the marine environment, thus preserving the services that healthy and resilient marine and coastal ecosystems provide (COM 2012). In addition to the traditional sectors of the blue economy (fisheries, oil and gas, shipbuilding and ship repair, and ferry and cargo transport), the strategy identified five areas for the development of blue growth: blue energy, aquaculture, coastal and marine tourism, blue biotechnology and seabed mineral resources.

Implementation of the Blue Growth Strategy is linked with other initiatives including the Marine Strategy Framework Directive (MSFD) and sea basin strategies such as the Maritime Strategy for the Atlantic Ocean Area (COM 2011) and the EU Biodiversity Strategy 2020 as noted above (Johnson, Ferreira et al. 2017). The MSFD is considered the environmental pillar of the IMP and represents an

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[7] See https://biodiversity.europa.eu/maes/mapping-ecosystems/indicators-for-ecosystem-services-marine for the list of indicators.

ecosystem-based approach to marine management (Mulazzani and Malorgio, 2017). The directive aims to protect the resource base upon which marine related economic and social activities depend. Included in the objectives is an analysis of the goods and services provided by the marine environment as well as the costs of degradation from anthropogenic activities (Mulazzani and Malorgio, 2017). These policies can all be linked through the scope of MSP, and indeed the IMP identified MSP as the cross cutting policy tool that would enable public authorities and stakeholders to apply a coordinated, integrated and trans-boundary approach to achieving sustainable growth and development of seas and oceans whilst maintaining ecosystems and achieving GES (COM 2014).

The Maritime Strategy for the Atlantic Ocean Area presents five objectives consistent with the "overriding objective of creating sustainable jobs and growth" of EUROPE 2020:

- implementing the ecosystem approach;
- reducing Europe's carbon footprint;
- sustainable exploitation of the Atlantic seafloor's natural resources (marine raw materials);
- responding to threats and emergencies; and,
- socially inclusive growth.

The Atlantic Action Plan (COM 2013) contributes to the Blue Growth strategy aiming to support the marine and marine economy in the Atlantic Ocean area. Its objectives, among others, are to drive forward the blue economy while preserving the environmental and ecological stability of the Atlantic Ocean. The plan encourages member states to cooperate in both traditional activities such as fisheries as well as emerging industries such as biotech and offshore renewables, while also preserving the environmental and ecological stability of the Atlantic.

There are however potential tensions between the different policy objectives. The recognition of the "overriding" nature of the priority of growth and jobs, while simultaneously recognising the need for "sustainability" and "social inclusion" is a recurrent thread in EU policy and indeed more generally. In Europe, the Cardiff process set the stage for mainstreaming environment concerns across European policy (COM 1998), and the Sustainable Development Strategy put emphasis on balancing the three pillars of sustainable development (Gothenburg Presidency Conclusions; European Council 2001). But in practice, the Lisbon Strategy (Lisbon Presidency Conclusions; European Council 2000) focus on '*sustainable economic growth with more and better jobs and greater social cohesion*' has been more emphasised in European policy (Hey 2005).

Nevertheless, the current ambition of transition to a Green Economy – and for the marine environment, Blue Growth – at least offers the hope of reconciling environmental objectives with economic and employment goals (Tinch et al 2015). The Marine Spatial Planning Directive, 'a cornerstone of the Commission's Blue Growth strategy', was described by Commissioners Damanaki's and Potočnik's joint statement as '*an important step in creating new growth opportunities across all maritime sectors by better managing our seas and ensuring their sustainability. Only if we coordinate the various activities taking place in our seas can we make access to maritime space more predictable for investors and at the same time reduce the impact of maritime activities on the environment*'[8]. Although the focus on growth is present, the need for sustainability and reduced environmental impact is also recognised. The priorities of the new Commission[9] put a "European Green Deal" front and centre, and the renewed focus on environmental sustainability under President Von der Leyen would seem to present a great opportunity for improving the MSP process. But this will only be possible with good knowledge of what the marine ecosystem services are, and how they will be impacted by changes in the economic activity taking place, in order for decision-making processes regarding the best use of those resources to ensure blue growth that is genuinely sustainable (Norton et al, 2018).

## 2.3 Climate change and connectivity

Currently, one of the greatest threats to deep-sea ecosystems is from on-going climate change (Johnson et al 2018) – sustainable marine management in general, and MSP in particular, need to be considered in the context of adapting to rapidly changing conditions. The ongoing rise of greenhouse gas concentrations is pushing marine systems toward conditions that are new, or at least not experienced for millions of years, and the rates of change are rapid, creating risks of major and perhaps irreversible ecological transformations (Hoegh-Guldberg & Bruno 2010). The changes include geochemical changes, with reduced pH and carbonate saturation levels unprecedented in the last 2 million years (Hönisch et al., 2009). Furthermore, although the expected timing varies according to emissions pathways and mitigation measures; the severe changes are not merely "High End Scenario" risks, but are expected under all four IPCC Representative Concentration Pathways (RCPs) (Hartin et al 2016). Sooner or later, therefore, these processes are likely to impact on habitat integrity and

[8] http://europa.eu/rapid/press-release_IP-14-459_en.htm

[9] https://ec.europa.eu/commission/sites/beta-political/files/political-guidelines-next-commission_en.pdf

representativeness, leading to changes in species ranges, and consequent changes in community composition and interactions (Pecl et al 2017, Roberts et al 2017). Though data are lacking, it seems likely that this will have severe impacts on deep-sea habitats and their fauna, through rise in $CO_2$ levels and ocean acidification, temperature change, expansion of hypoxic zones, destabilization of the slopes and gas hydrates and changes in productivity regimes; benthic communities on sedimentary upper slopes, cold-water corals, canyon benthic communities and seamount pelagic and benthic communities are particularly at risk (Ramirez-Llodra et al 2011).

Climate change also interacts with marine connectivity. Most obviously, climate-related disturbances can change larval dispersal via impacts on hydrodynamics as well as on biological factors including spawning times, pelagic larval durations, larval mortality and behaviour (Magris et al 2014). Different life stages may be differently sensitive to temperature, and adult spawning fishes may be especially sensitive (Asch & Erisman, 2018). At the same time, connectivity influences how (meta-)populations could respond to climate shocks and events. For example, increased abundance of marine species in marine protected areas (MPAs) is expected to enhance productivity of the surrounding areas which can help buffer against climate impacts and increase resilience (Roberts et al., 2017).

These considerations could have complex implications for marine conservation, for example if reduced larval durations enhanced larval survival but reduced connectivity. In one study, higher SST resulted in reduced planktonic duration but increased egg and larval mortality. Results predict reduced dispersal to more distant sites but increased dispersal to neighbouring sites. Increased mortality modified the magnitude of population connectivity but had little effect on the overall patterns (Young et al 2018). Hence, species connectivity patterns that track climate change must be considered in MSP and specifically in the design of robust MPA networks (Carr et al 2017). Overall, climate change threatens the effectiveness of MPAs (Bruno et al 2018), and designing MPA networks without taking these impacts into account "could result in major investments being made in areas that will not survive the next several decades" (McLeod et al 2009).

To achieve this, the spatial connectivity and potential sensitivity to climate change must first be determined. This could include attempting to identify critical areas that are most likely to survive the threat of climate change (McLeod et al 2009): responses to climate change and large-scale forcing can vary widely at local scales creating marine microclimates that can be robust even under extreme large-scale forcing events (ENSO, climate change) potentially creating spatial refuges or 'safe spaces' for important species (Woodson et al 2018). Other options include seeking to compensate for the increased mortality effect through increasing the size or number of protected areas in order to increase populations and reproductive output, or adding additional nodes to MPA networks to adapt to reduced distances of larval dispersal: regionally networked marine reserves can provide stepping-stones for dispersal, safe "landing zones" for colonizing species, and possible refugia for those unable to move (Roberts et al 2017).

All these changes will inevitably influence the effectiveness of existing (spatial) management measures, such as networks of MPAs, and the spatial distribution, provision and value of marine resources and ecosystem services. At present, we lack the biological knowledge and volume of data to predict these responses in a consistently reliable manner. Nevertheless, MSP is an urgent priority and must make the best use possible of such knowledge and data as are available. Increasingly, this knowledge base is expanding to include information about the values to humans of marine ecosystem services, now and in the future.

# 3 Atlantic Ocean values

Prior to presenting an assessment of the valuation knowledge base, it is expedient to address the question of what is meant by value in this context, in particular because one common problem in the use of monetary values is confusion between different interpretations of the value concept. We then examine the availability of valuation evidence, and consider how it might be used in MSP. We conclude with an assessment of progress being made in ATLAS towards providing the economic baseline needed for effective MSP.

## 3.1 Valuation frameworks

Economics is founded on the principle that "scarcity implies choice" (Robbins, 1935); it is not possible to achieve all objectives simultaneously, so trade-off, whether implicit or explicit, is inevitable (Costanza *et al*., 2011). Economic valuation is concerned with assessing trade-offs in explicit terms (Farber et al., 2002). Burkhard et al (2013) argue that while multiple indicators can be used for assessing trade-offs across different ecosystem services, the analysis can be easier if these are standardised to a common indicator, generally economic value.

However, value can mean different things in different contexts, even within the relatively narrow confines of economic analysis. In economics, two main analytical frameworks are common:

- Exchange values
    - Based on trade in markets, focusing on the price of exchange and the quantity traded.

    - Commonly used to assess economic impacts in terms of changes in gross value added (GVA), often alongside assessment of impacts on jobs/employment.
    - Underpins economic impact assessment (EIA), used to estimate changes in levels of economic activity within a specific area.
    - Underpins the national accounts of gross domestic product (GDP)
    - Similar methods/metrics are now being used for environmental and ecosystem accounting, which focuses on exchange values to maintain comparability with the national accounts.
- Total economic value (TEV)
    - Based on how much individuals are/would be willing to pay for a given good or service, or how much they would be willing to accept as compensation for giving it up.
    - The foundation for human welfare-based assessments of value and the cost benefit analysis (CBA) framework.
    - Less commonly used in marine settings due in part to data/knowledge gaps.
    - Potentially important since it is capable of taking account of impacts on ecosystem services via non-market valuation techniques/evidence.

Both frameworks look at impacts in different ways, with different assumptions, strengths and weaknesses. If the main interest is in effects on national income and employment, then exchange values and EIA are appropriate. If interest is rather in impacts on social welfare, TEV and CBA are appropriate. But these are not mutually exclusive interests – on the contrary, decision makers and stakeholders will often wish to consider both, and both have a role to play in MSP. To date, however, EIA is more common, while TEV lags behind, in particular in marine settings, partly due to data and valuation evidence gaps. This leads to a risk of focusing too narrowly on market impacts, at the expense of the wider set of goods and services that influence human wellbeing and environmental sustainability.

There may be a particular risk here with regards to the current drive towards greater use of ecosystem accounting, since the focus there on using exchange value concepts can result in low values being used for non-traded ecosystem services while simultaneously giving the impression that the services have nevertheless been "taken into account". This can be misleading, since "the valuation methods that are consistent with accounting only aim to quantify ecosystems contribution to the economy, not societal well-being or welfare" (Grimsrud et al 2018). As a further complication, this criticism applies specifically to environmental and ecosystem accounting in the UN's SEEA framework, and not so much to wealth accounting as in the WAVES approach which is aligned with the TEV framework (but therefore not appropriate for comparison with the national accounts and derived indicators).

Ecosystem service valuation in the TEV framework can highlight the 'hidden' ecosystem benefits and costs. This in turn can improve understanding of the economic trade-offs from different marine plans or scenarios, including trade-offs among different kinds of ecosystem services, as well as between ecosystem services and the commercial economic activities that impact on the condition of marine ecosystems (Börger et al 2014). For the deep seas, the TEEB Oceans study (TEEB 2012) present two important questions to consider in this regard:

- What economic information do we need to weigh the trade-offs between the industrial exploitation of the deep sea and the emerging economic value of living resources there?
- Can a better economic understanding of the value of deep-sea ecosystems help in the design of industrial best practices, deep sea marine protected areas, and international governance of the deep sea?

There is increasing acceptance that, despite the serious challenges of valuing deep-sea ecosystem services, valuation evidence could indeed help improve decision making. The need to enhance and facilitate integration in decision making is widely recognised and a focus of recent effort – for example the Ecosystem Services Partnership (ESP) Regional Europe conference in 2018 on the theme "Ecosystem services in a changing world: moving from theory to practice"[10] - although, illustrating the challenge, the conference was mainly attended by the research community. Nevertheless momentum is gathering. The European Parliament Intergroup Seas, Rivers, Islands and Coastal Areas (Searica Intergroup) together with the European Marine Board organised a conference[11] to discuss requirements for assessing the long-term sustainability of blue growth, support ecosystem based policy development and marine management decisions, and raise awareness of the importance of the marine environment to society and in the economy. The European Commission Blue Economy Report 2019[12] includes a chapter on ecosystem services and natural capital, noting increasing evidence on the value of ecosystem services. They report that the few cost-benefit studies available conclude that the overall welfare benefits of marine protected areas are positive, but that with a narrower focus on the market impacts in Blue Economy sectors, the situation is less clear. This illustrates the importance of being able to account for the non-market improvements in societal welfare (in particular, using valuation in the TEV framework for CBA), because the market impacts alone may not be enough to offset the costs of action.

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[10] https://www.espconference.org/eu2018#.Ws3KoDN7E3g

[11] "*Valuing Marine Ecosystem Services - Taking into account the value of ecosystem benefits in the Blue Economy*" Brussels, April 2019

[12] https://prod5.assets-cdn.io/event/3769/assets/8442090163-fc038d4d6f.pdf

For the ABNJ/BBNJ, where governance structures are weak or incomplete, there may be additional roles for expanding the knowledge base on economic values and ecosystem services. These include, for example (TEEB 2012): to inform new agreements to implement EBM and MSP; increasing the effectiveness of tools such as MPAs, EIA, and SEA; designing benefit sharing mechanisms for marine genetic resources beyond national jurisdiction; and providing incentives for transfer of technology and innovative financial mechanisms for capacity development and implementation.

Enhancing the valuation evidence base and integrating values in MSP is also seen as a way to encourage investment and blue growth. For example, the Mid-Atlantic Regional Ocean Assessment (ROA) reports[13] ongoing research to investigate the suitability of modelling tools based on ecosystem services valuation, including Marine InVest,[14] exploring the potential for marine-focused market mechanisms for ecosystem services such as water filtration provided by oyster beds, and wave attenuation and carbon sequestration services of tidal marshes. In Europe, the ALICE project[15] has the main goal of promoting sustainable investments in Blue-Green Infrastructure Networks (BGINs) through identification of the benefits of Ecosystem Services delivered at the terrestrial-aquatic and land-sea interface in the Atlantic Region.

## 3.2 Ocean goods and services

There are a number of different definitions for ecosystem services. Ecosystem services can be defined as '*the benefits that people obtain from ecosystems*' (MA 2005) or '*the direct and indirect contributions of ecosystems to human well-being*' (TEEB 2010) or *'marine ecosystem services are provided by the processes, functions and structure of the marine environment that directly or indirectly contribute to societal welfare, health and economic activities'* (Norton et al, 2018; Austen et al, 2019). Identification of services associated with the deep sea aids decision makers to focus their attentions on the best initiatives to protect deep sea ecosystems while also safeguarding commercial interests, livelihoods and societal values (ATLAS D5.1) and facilitates ecosystem service valuation (Austen et al, 2019).

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[13] *https://roa.midatlanticocean.org/ocean-ecosystem-and-resources/characterizing-the-mid-atlantic-ocean-ecosystem/ecosystem-services/*

[14] https://naturalcapitalproject.stanford.edu/invest/

[15] *http://project-alice.com/alice-project/*

Frameworks for the identification and classification of ecosystem services have evolved over the years in particular since the publication of the Millennium Ecosystem Assessment (MA) (MA 2005; Tinch et al. 2011). Among these are The Economics of Ecosystems and Biodiversity (TEEB), the UN Common International Classification of Ecosystem Services (CICES) and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (TEEB 2010; CICES 2013; IPBES 2017). Such frameworks have been developed to help differentiate, give structure to and provide the basis to evaluate ecosystem services (Thurber et al. 2014). The following categorisation of ecosystem services was proposed by the Millennium Ecosystem Assessment (2005) and forms the basis of most other classification systems (Costanza et al. 2017).

- ***Provisioning Services*** are the products used by humans that are obtained directly from the ecosystem for example commercial fish
- ***Regulating Services*** are the benefits obtained through the natural regulation of ecosystem processes such as gas and climate regulation, and carbon sequestration
- ***Cultural Services*** are the often non-material benefits people obtain from ecosystems through recreation, aesthetic environment, 'inspiration' and 'awe'
- ***Supporting Services*** are those functions and processes that are necessary for the production of all other ecosystem services, i.e. they feed into provisioning, regulating and cultural services thus feeding *indirectly* to human wellbeing.

*Figure 1* presents the ecosystem services identified for the deep sea using the MA framework (adapted from Armstrong et al, 2012).

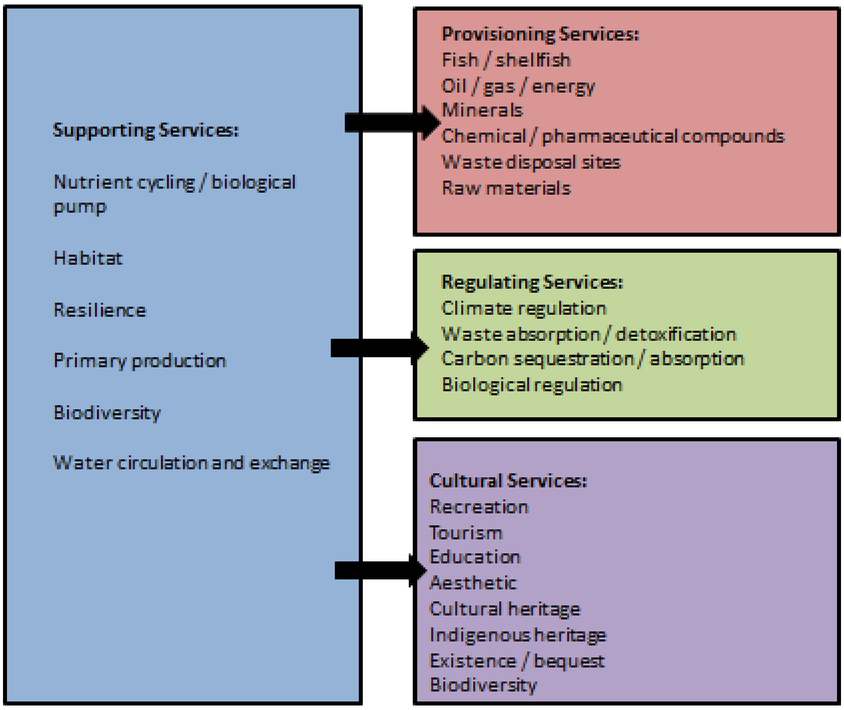


*Figure 1: Ecosystem Services in the deep sea, using the Millenium Ecosystem Assessment fromework*

The Common International Classification of Ecosystem Services (CICES) takes a slightly different approach, excluding supporting services (and also biodiversity) from valuation[16]. The rationale (Haines-Young and Potschin, 2013) stresses a clear distinction between final ecosystem services and ecosystem goods or products:

- Human well-being arises from adequate access to the basic materials, freedom of choice and action, health, good social relations and security. This is partly dependent on access to ecosystem goods and benefits.
- Ecosystem goods and benefits are created or derived from final ecosystem services by humans. These products and experiences "are no longer functionally connected to the systems from which they were derived."
- Final ecosystem services, in contrast, retain a direct connection to the underlying ecosystem functions, processes and structures that generate them. They are 'final' as the outputs of ecosystems that most directly affect human well-being. CICES is a classification at this level i.e. services, not benefits.

[16] http://cices.eu/

- Intermediate and supporting services are functions and processes that underpin the final services. They are not directly included in CICES because they are only indirectly consumed or used, and may simultaneously facilitate the output of many 'final outputs'.

Thus the exclusion of supporting services from CICES is not intended to suggest that they are unimportant. Rather, the rationale is directly connected to accounting: "if ecosystem and economic accounts are to be linked, then an essential step is to identify and describe the 'final outputs' from ecosystems that people use and value, so as to avoid the problem of double-counting" (Haines-Young and Potschin, 2013 p8). Though at the same time "there is no reason why fully developed environmental and economic accounts cannot also record changes in underlying ecological structures, processes and functions, and systems like CICES may well be extended to cover them" (ibid, p8) – but in physical terms, not monetary, to avoid double counting. In other words, CICES is intended to provide a framework focused on final services within which information about supporting or intermediate services can be nested and referenced. Haines-Young and Potschin (ibid) argue that such treatment may be especially useful for mapping ecosystem services and propose that "CICES should be explored through the development of experimental accounts, especially in the context of using accounts to check the integrity of underlying ecological assets" (ibid, p8).

In the USA, an alternative framework has been developed: the FEGS-CS (Final Ecosystem Goods and Services Classification System). This is in some respects similar to CICES, in that it also focuses on final goods and services, but has an additional emphasis on classifying both service and beneficiary together, rather than focusing just on the service as an ecosystem feature.

There is no single best way to classify ecosystem services, and the frameworks have evolved over the years, with the final selection depending on the ecosystem and policy context (Tinch et al. 2011). The main evolution in the ES frameworks from the MA from a valuation perspective is that they focus on the direct services, largely excluding the indirect supporting services from this step. The motivation for excluding supporting services is to avoid the issue of double counting ecosystem services in valuation (ATLAS D5.1).

Specifically for the deep seas, recent work has started to study the relationship between ecology and ecosystem service, notably Danovaro et al (2008a,b). Applications of ecosystem services concepts in marine contexts span a short history (Liquete et al., 2013). Thurber et al (2014) argue that ecosystem services frameworks developed for terrestrial environments are not well suited to the deep sea due to the low resolution of spatially explicit marine information and the difficulty of quantifying ecosystem functions and processes in the highly dynamic and connected three-dimensional marine environments. Indeed many of the services identified by Armstrong et al (2012) are supporting or intermediate services in the deep sea that underpin crucial final services elsewhere in space and time. This does not sit well with recent approaches such as CICES (Haines-Young & Potschin 2013) or FEGS (Landers & Nahlik 2013) which focus only on the final services, as explained above. Le et al (2016) meanwhile stress the likelihood of discovering previously unknown final and supporting services. The complexity of marine ecosystems, and their connectivity with other systems and services across space and time, makes knowledge transfer very challenging (Jobsvogt et al, 2014b).

Hence, to present the role of the deep sea for human wellbeing in a transparent way, insofar as our knowledge allows this, ecosystem functions or supporting services need to be described (Armstrong et al. 2010). This is particularly true of the deep sea, since many of the final services supported by deep-sea functions create values that are distant in space and time from the deep sea, and may fall outside the spatial or temporal boundaries of the specific assessment or plan area. It is essential under such circumstances to consider the supporting services of the deep sea that maintain the ability of the other systems to provide final services (Tinch et al. 2011). Furthermore, the ATLAS expert assessment on risks (ATLAS D5.2) found that supporting services were perceived to be most at risk. If the focus is only given to the three service types that impact humans directly, important impacts and risks may be ignored, particularly for the deep sea (Armstrong et al, 2019a). The MA, though somewhat dated in certain respects, is useful to describe services in the deep sea as it includes supporting services. The approach adopted by ATLAS is therefore to draw on two frameworks: the first is to describe a broad set of the services in the deep sea using the MA. The second uses CICES to inform the monetary valuations, and thereby avoid the issue of double counting. Nevertheless, care is required in interpreting the results of these valuations to ensure that important services arising outside the marine environment, but supported by it, are not overlooked.

## 3.3 Assessment of evidence base

From the perspective of ecosystem services valuation in general, a growing number of original economic valuation studies, meta-analyses of economic valuation studies (e.g. Brouwer et al., 1999; Brander et al., 2011) and economic valuation databases[17] has consolidated the evidence base and

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[17] See in particular the Environmental Valuation Reference Inventory (EVRI) (www.evri.ca), the TEEB valuation database (http://es-partnership.org/services/data-knowledge-sharing/ecosystem-service-valuation-database/; de

facilitated the transfer of economic value estimates to new contexts (such transfer being considered contentious by some: see e.g. Ravenscroft, 2019). The mainstreaming of economic valuation is demonstrated by the development under the environmental management systems series International Standards Organisation (ISO) 14000 (the best-selling standard in the world) of ISO 14007 "Environmental management: Determining environmental costs and benefits – Guidance"[18] and ISO 14008[19] "Monetary valuation of environmental impacts and related environmental aspects".

Nevertheless, economic valuation of deep marine ecosystems remains particularly challenging and understudied. The consequent risks of incomplete evidence and undervaluation make it particularly important that both qualitative and quantitative narrative is presented alongside any monetary assessment.

### 3.3.1 Values in literature

Ecosystem services valuations have mostly been developed and implemented for terrestrial ecosystems, with their application lagging behind in the marine environment, mainly due to lack of data (Beaumont et al, 2019). Knowledge of marine ecosystem services and their socioeconomic values is therefore limited (Armstrong et al. 2012), although there is increasing interest in identifying and estimating these services and values. Attention is largely focused on coastal areas (de Groot et al. 2012; Liquete et al. 2013; Beaumont et al. 2014), which are more familiar, more heavily used, and closer to human populations. In tropical areas, people can be especially dependent on coastal ecosystem services, as well as exposed to ecosystem-related risks, and this has been a particular priority for valuation research (de Groot et al 2013). Nevertheless, although clearly less studied than terrestrial, fresh water and coastal environments, there has been increasing recognition of the importance of the services provided by the deep sea (Tinch et al. 2011). van den Hove and Moreau (2007) discuss the socio economics of the deep sea, including ecosystem services, as well as the impacts and pressures the deep-sea environment faces from human activities. Armstrong et al (2010; 2012) build on this work, presenting a categorisation and synthesis of deep-sea ecosystem goods and

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Groot et al., 2012), the Envalue database (http://marineecosystemservices.org/explore) and the Marine Ecosystem Services Partnership's (MESP) Valuation Library (http://marineecosystemservices.org/explore).

[18] https://committee.iso.org/sites/tc207sc1/home/projects/ongoing/iso-14007.html

[19] https://committee.iso.org/sites/tc207sc1/home/projects/ongoing/iso-14008.html

services, reviewing the state of knowledge regarding these services and possible methods for their valuation. Thurber et al (2014) provide further discussion on deep sea ecosystem services and functions, identifying traits that differentiate the deep-sea habitats from other global biomes.

Applied valuation studies to the deep sea and associated ecosystems include discrete choice experiments (Glenn, Wattage et al. 2010; Wattage, Glenn et al. 2011; Jobstvogt, Hanley et al. 2014; Aanesen, Armstrong et al. 2015), contingent valuation surveys (Ressurreição, Gibbons et al. 2011; Ressurreição, Gibbons et al. 2012; Ressurreição, Zarzycki et al. 2012) and benefit transfer (Beaumont et al, 2008). Norton et al (2018) provide an overview of ecosystems services in Irish waters including deep sea. Some deep-sea ecosystems have received more attention than others: for example Foley et al (2010) identify the ecological goods and services associated with cold water coral ecosystems and there have been several attempts to value CWC systems (see below). Overall, however, it remains that deep-sea habitats receive less attention than environments closer to home due to their remoteness, unfamiliarity, and difficulty to access (Foley et al, D5.1).

Consequently, and indeed to an even greater extent, monetary values from deep sea ecosystem services are sparse compared to terrestrial and coastal environments (see Table 1). Existing information is usually tied to the provisioning services of the ocean such as fisheries and fish habitat; with little information on regulating and cultural services, or future potential services from Blue Growth. Provisioning services such as fisheries are quantifiable, but regulating or cultural services are not well known to the public. This makes total valuation a demanding exercise, but one that has been attempted for a few deep-sea ecosystems, such as cold-water corals.

The few studies that have been carried out demonstrate that people are willing to pay for protection of deep-sea ecosystems, despite their remoteness and lack of familiarity. A recent example (Norton and Hynes 2018) used a combination of the contingent valuation and value transfer to estimate the value of non-market benefits associated with the achievement of GES as specified in the MSFD for Atlantic member states. The study estimated that the overall value of achieving GES for five Atlantic member states varied between €2.37 billion and €3.64 billion.

Other examples include a discrete choice experiment for the deep sea area of the north and northwest UK EEZ to value both use and non-use values attached to deep-sea environments around the Scottish coast (Jobvogt, 2014). The study found a WTP for deep sea protection ranging from £70 - £77 despite the remoteness and lack of familiarity with the areas. Aanesen et al. (2015) designed a DCE to derive willingness to pay for increasing the protection of cold-water corals in Norway. Choice attributes were selected using existing literature and expert interviews. The possibility that CWC play an important role as a fish habitat was the most important variable to explain people's WTP for the protection of CWC. The study found a high WTP for the protection of CWC in the range of €274 - €287. Ressurreição et al. (2011) use a contingent valuation method to estimate the public's WTP to avoid loss in the number of species in the marine waters around the Azores. The aim of the study was to estimate the marginal value associated with increased levels of species loss, and also to estimate the WTP to avoid loss of species in different marine taxa. The results suggested a greater WTP to preserve all marine taxa as a whole, than for a series of individual marine taxa.

These studies provide a proof of concept for applicability of valuation techniques to deep sea environments. However the body of evidence remains thin, and not fully adequate for the task of supporting the development of MSP for the North Atlantic. To help fill this gap and to add to the economic knowledge base, ATLAS has carried out a number of economic valuation studies, as reported in full in Deliverable 5.4 and Deliverable 6.2, and as discussed further below.

## 3.4 ATLAS case studies

ATLAS is testing a generic MSP framework developed by the FP7 MESMA project to assess spatially managed areas (SMAs) in all 12 of the ATLAS Case Studies.

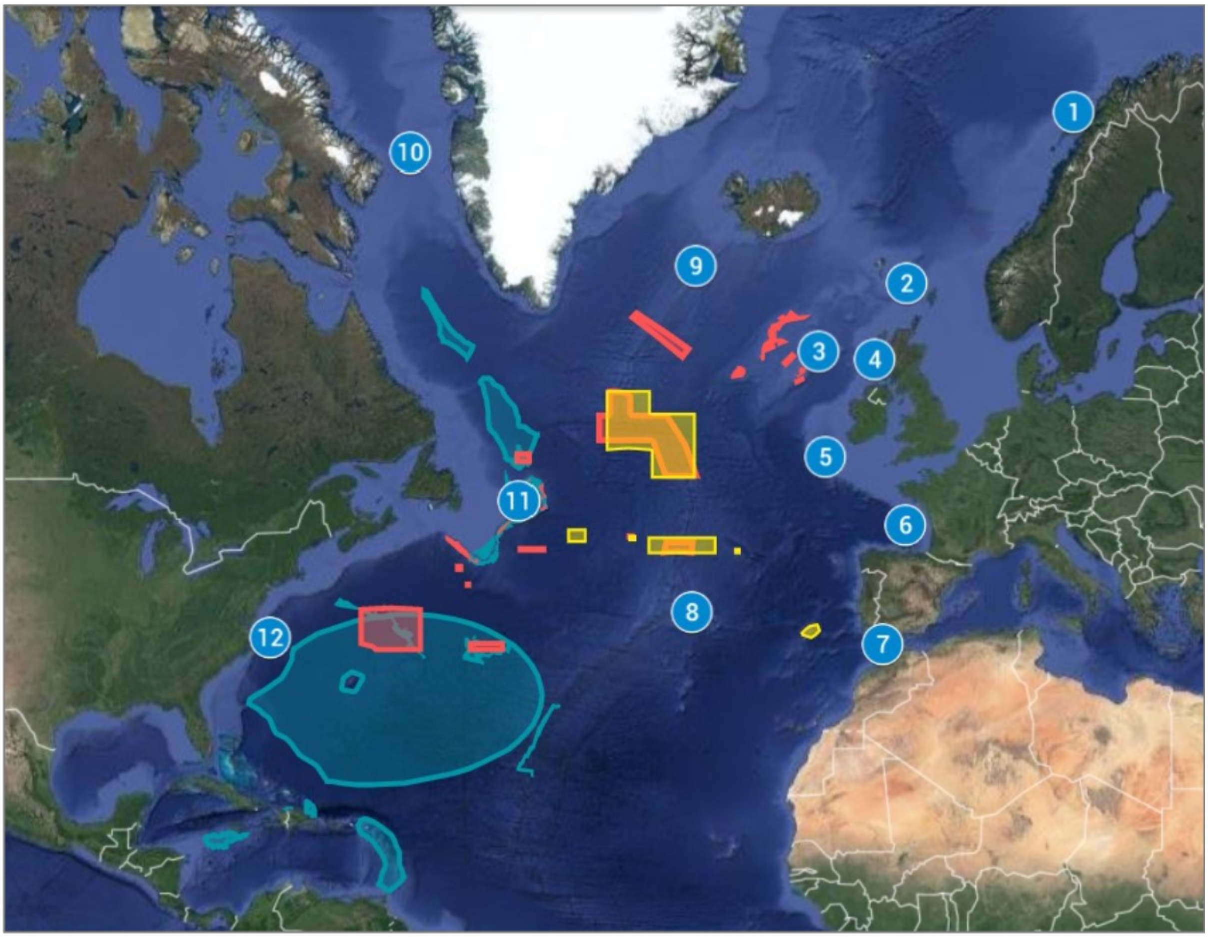


*Figure 2: ATLAS Case Study locations (numbered items) overlaid with Ecologically or Biologically Significant Areas (light blue areas); Vulnerable Marine Ecosystems (red boxes) and OSPAR Marine Protected Areas in Areas Beyond National Jurisdiction (yellow boxes). (Source: Grehan et al 2018 / D6.1).*

The relative lack of data available for deep sea benthic ecosystems relative to terrestrial or even near shore benthic ecosystems constrains the possible assessment of ecosystem services. ATLAS therefore uses a mixed methods approach, with a qualitative approach to map the expected or potential ecosystem service delivery levels initially for 12 ecosystem service types (Galparsoro et al 2014) and a quantitative approach for estimating the ecosystem service of food provision through generated estimates of landings volume and value for case studies. This has been enhanced by use of a Delphi survey (ATLAS D5.2) to assess expert views on the relationships between human activities and ecosystem services, and through choice experiments as discussed below.

### 3.4.1 ATLAS case studies: risks

One way to consider the balance between the blue growth economic agenda and sustainability is to assess the potential impacts or risks posed by different forms of human activity on the ecosystem services provided by the deep sea. Such an assessment of impacts and risks will also inform MSP (Armstrong et al, 2019a). ATLAS D5.2 carried out a Delphi study to assess risks to ecosystem services in the North Atlantic Ocean from climate change, other human stressors and their cumulative effects (Armstrong et al, 2019a). Human drivers perceived to pose most risk to ecosystem services in the Atlantic deep sea are pollution, ocean acidification, fisheries and cumulative effects. The services most impacted are the provisioning service of fish and shellfish, and supporting and cultural service of biodiversity, as well as supporting service of habitats (Figure 3).

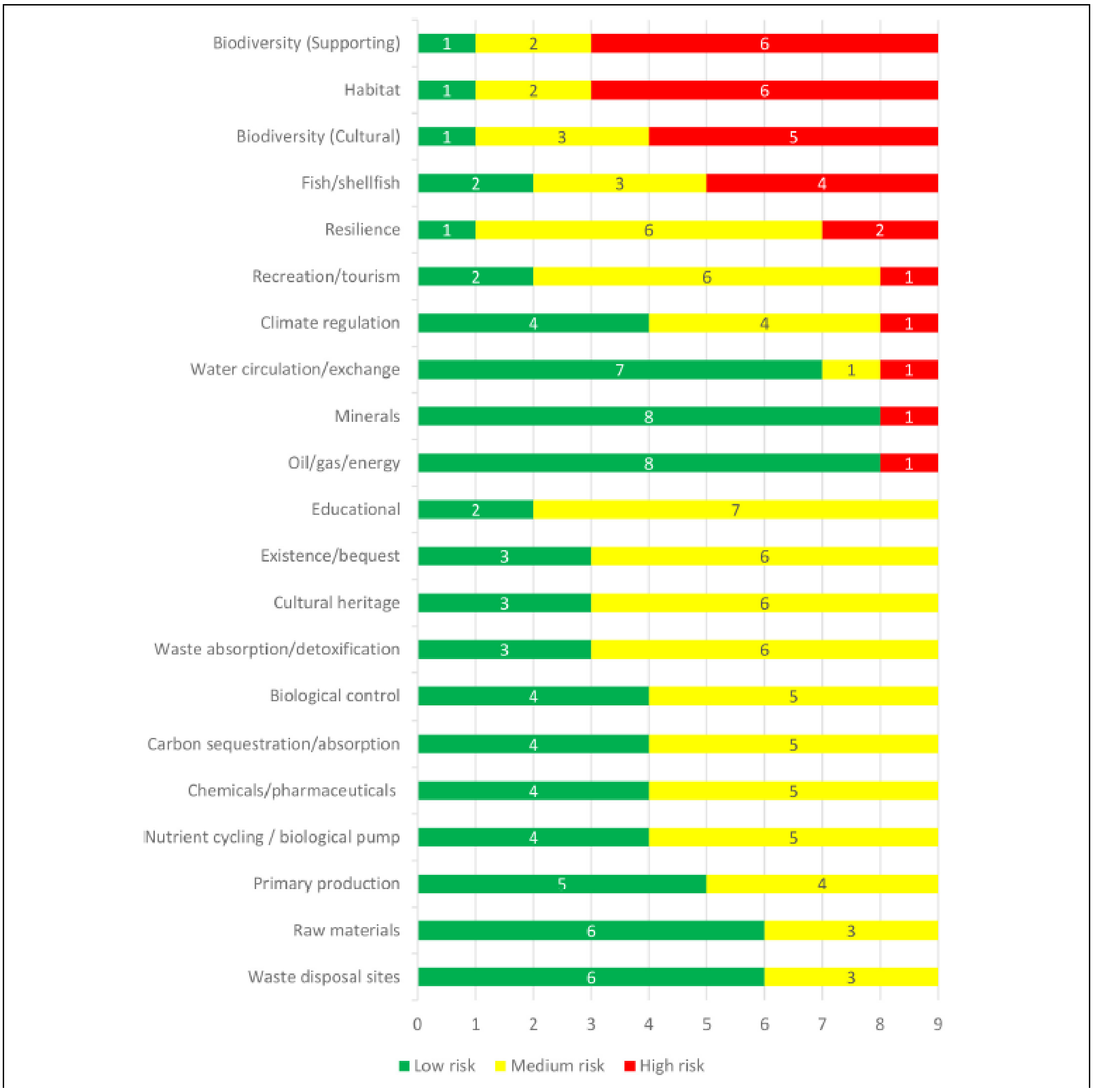


*Figure 3: Ecosystem service risk levels arising from human drivers. The horizontal axis represents the number of drivers within each risk category.*

### 3.4.2 ATLAS case studies: Inventory of Ecosystem Services

**ATLAS** Deliverable 5.1 carried out an inventory and qualitative review of ecosystem services for each of the case study areas. As discussed above many different classification systems exist for ecosystem services. Recognising the importance of supporting services to the deep sea but also the issue of double counting in valuation two frameworks were used – the Millennium Ecosystem Assessment and CICES – to present the services. The data were collated through a survey, discussion with scientists and a literature review of each case study area.

Despite the many unknowns regarding the deep sea, a major outcome of the inventory was the large number of services identified for each area. An overview of the ecosystem services identified for each case study area is presented in table x. The coloured cells indicate that the ecosystem service was identified as present for the particular case study area. The colours represent each service type – supporting, provisioning, regulating and cultural. The table also indicates using symbols whether the service is present (+); monetarily known (€); not present (0) and unknown (?) following earlier work of Armstrong et al (2012) and Beaumont et al (2007). Monetary values are mainly present for provisioning services in particular fisheries and some values deemed present for cultural services where valuation studies have been carried out or activities such as tourism indicated as present in current commercial activities. For the most part, Table 1 shows that there are a lot of services known to be present in the case study areas but for most, the monetary values have not yet been established.

There has also been a number of interesting qualitative assessments of deep-sea ecosystem services that can feed into MSP. One study (Salomidi et al. 2012) created a sea-floor habitat ecosystem service scoring system drawing on a standardized system of classifying ecosystems through EUNIS[20] codes and a compilation of the goods, services, sensitivity, and conservation status of 56 European sea beds. A later development (Galparsoro et al. 2014) expanded this to 62 sea-floor habitats, qualitatively evaluating the ecosystem services. ATLAS has drawn on these methods in the analysis of service values for the case studies, as discussed below.

[20] http://unis.eea.europa.eu

| | ECOSYSTEM SERVICES | LoVe | Mingulay | Azores | Flemish Cap | West of Shetland and W of Scotland Slope | Rockall Bank | Porcupine Seabight | Bay of Biscay | Gulf of Cadiz/Strait of Gibraltar/Alboran Sea | Reykjanes Ridge | S Davis Strait/Western Greenland/Labrador Sea | SE USA (Bermuda Transect) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| SUPPORTING | Nutrient cycling/biological pump | + | + | + | + | + | + | + | + | + | + | + | + |
| | Habitat | + | + | + | + | + | + | + | + | + | + | + | + |
| | Resilience | + | + | + | + | + | + | + | + | + | + | + | + |
| | Primary production | + | 0 | + | + | 0 | + | + | + | + | + | + | + |
| | Biodiversity | + | + | + | + | + | + | + | + | + | + | + | + |
| | Water circulation/exchange | + | + | + | + | 0 | + | + | + | + | + | + | + |
| PROVISIONING | Fish/shellfish | € | € | € | € | € | € | € | € | € | € | € | € |
| | Oil/gas/energy | ? | + | + | € | € | + | € | 0 | + | + | € | + |
| | Minerals | 0 | 0 | + | + | + | + | + | 0 | + | + | + | + |
| | Chemical/Pharmaceuticals | 0 | + | + | + | + | + | + | 0 | + | + | 0 | + |
| | Waste disposal sites | | | + | + | + | | 0 | + | + | 0 | 0 | + |
| | Raw materials | + | + | 0 | 0 | 0 | 0 | | 0 | + | 0 | 0 | + |
| REGULATING | Climate regulation | + | + | + | + | + | + | + | + | + | 0 | + | + |
| | Waste absorption/detoxification | + | + | + | + | + | 0 | + | + | + | 0 | 0 | + |
| | Carbon sequestration/absorption | + | + | + | + | + | + | + | + | + | 0 | + | + |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Biological regulation | 0 | + | + | + | 0 | + | + | + | + | + | + | + |
| CULTURAL | Recreation | + | + | + | 0 | 0 | 0 | 0 | + | + | 0 | 0 | |
| | Tourism | + | + | + | 0 | + | 0 | 0 | 0 | + | 0 | 0 | + |
| | Educational | + | + | + | + | + | + | 0 | + | + | + | + | + |
| | Aesthetic | + | + | + | + | + | + | 0 | 0 | + | + | + | + |
| | Cultural heritage | + | + | 0 | + | + | + | + | 0 | + | 0 | + | + |
| | Indigenous heritage | 0 | + | 0 | 0 | + | 0 | 0 | 0 | + | 0 | € | + |
| | Existence/bequest | € | + | + | + | + | + | 0 | + | + | + | + | + |
| | Biodiversity | + | + | + | + | + | + | + | + | + | + | + | + |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ACTIVITIES | Fisheries | | | | | | | | | | | | |
| | Aquaculture | | | | | | | | | | | | |
| | Oil and gas | | | | | | | | | | | | |
| | Marine mineral mining | | | | | | | | | | | | |
| | Marine biotechnology | | | | | | | | | | | | |
| | Tourism | | | | | | | | | | | | |
| | Renewable energy | | | | | | | | | | | | |
| | Scientific Ref Sites / Observatory | | | | | | | | | | | | |
| | Shipping | | | | | | | | | | | | |
| | Cables | | | | | | | | | | | | |

*Table 1: Overview of the provision of goods and services in the case study areas. + present; € monetary values available; 0 not present. Adapted from Armstrong et al (2012) and Beaumont et al (2007). (Activities table presents current (orange) and blue growth potential (blue) per case study area)*

### 3.4.3 ATLAS case studies: mapping services

For mapping seafloor habitats in case studies, ATLAS used data from EMODnet (European Marine Observation and Data Network[21] and EUSeaMap (Mapping European seabed habitats[22]). *Figure 4* shows the extent of EU-SeaMap sea-floor habitat mapping for the main deep sea benthic EUNIS habitats found in the case study sites.

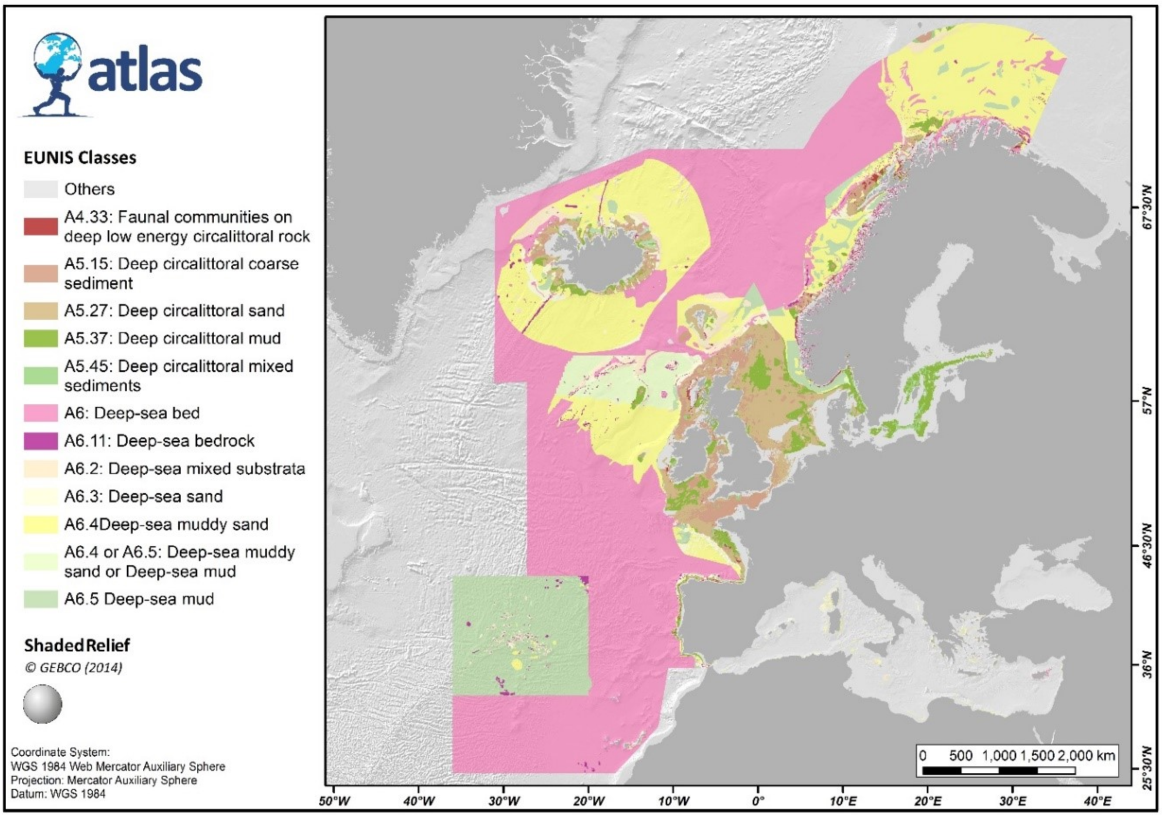


*Figure 4. EU-SEA Map 2016 with EUNIS Codes for sea-floor habitats in the ATLAS Cases Studies.*

Applying these methods leads to a spatial assessment of the relative values of 12 ecosystem services across the case study areas in ATLAS, as illustrated in the following examples (*Figure 5*, Figure 6). For fisheries, full monetary evaluation is possible (*Figure 7*). This remains however a static analysis; a fuller appreciation of the situation can be achieved via analysis of the risks posed to vulnerable marine ecosystems and fish habitats through fishing pressure (*Figure 8*). These are examples of analysis provided in full detail in ATLAS reporting (Deliverable 6.2).

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[21] http://www.emodnet-hydrography.eu/; European Commission; Directorate-General for Maritime Affairs and Fisheries (DG MARE)
[22] http://jncc.defra.gov.uk/page-6266

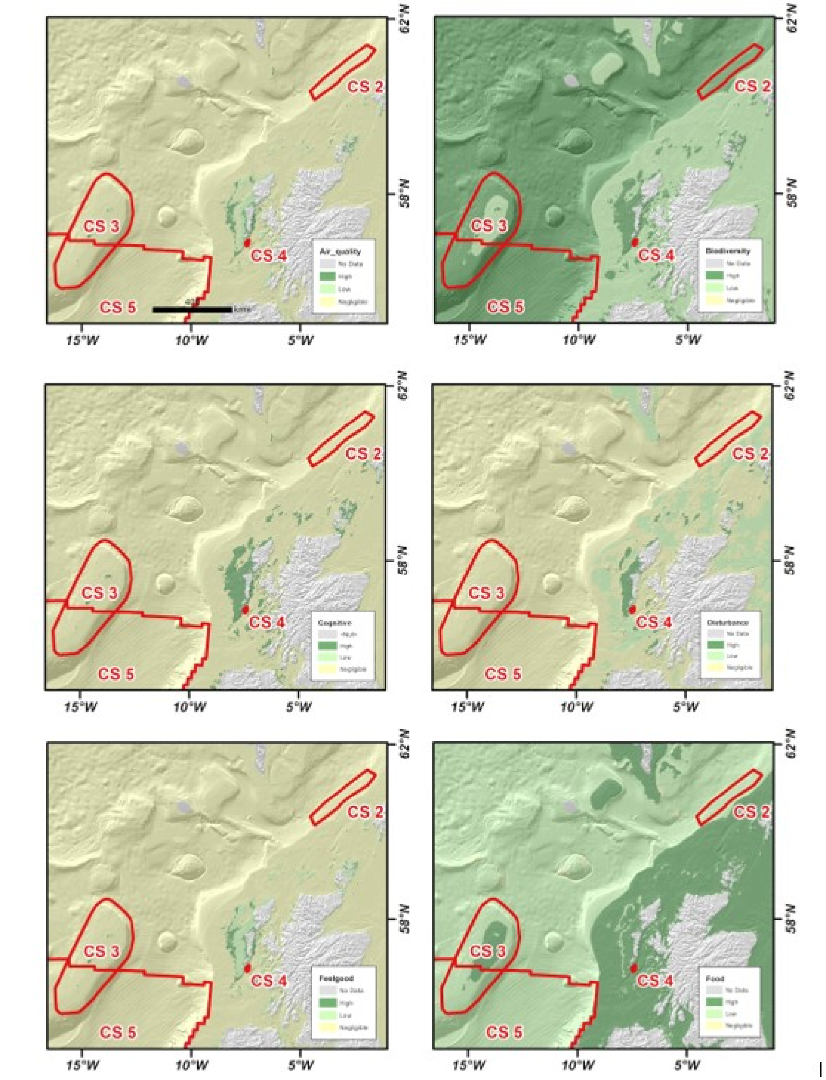



*Figure 5: Ecosystem service scores for Mingulay Reef, UK*

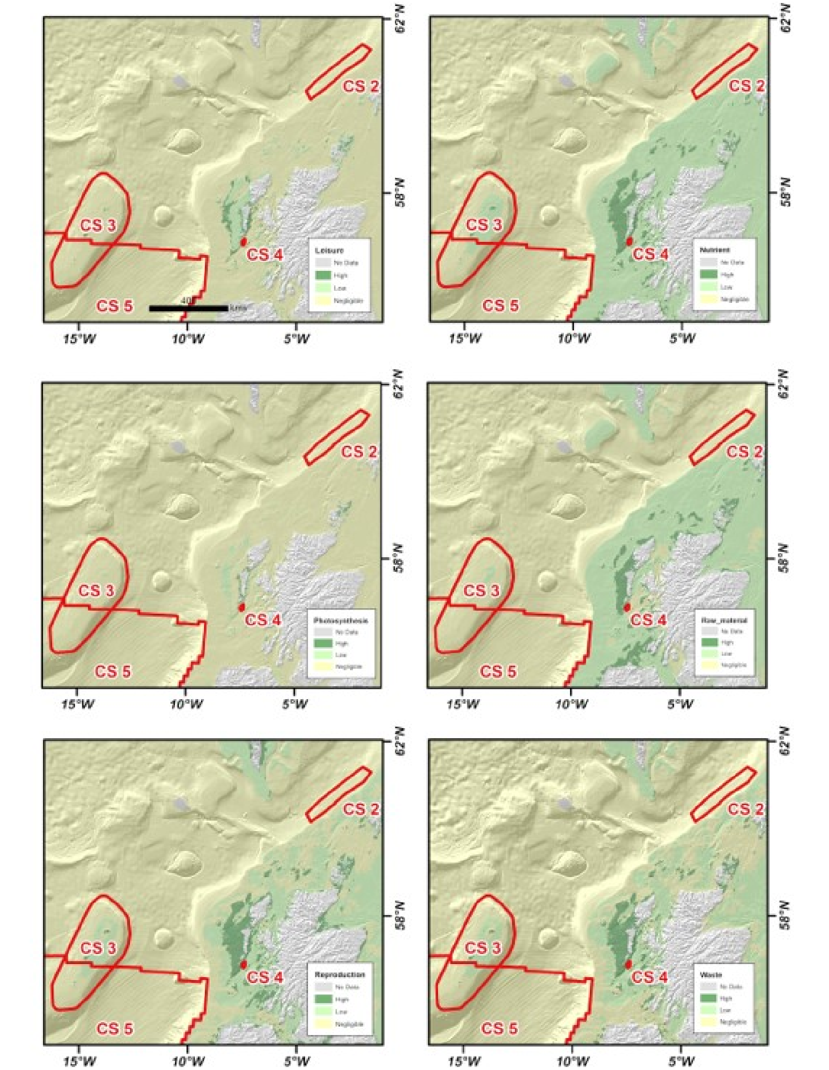



*Figure 6: Ecosystem service scores for Mingulay Reef, UK (continued).*

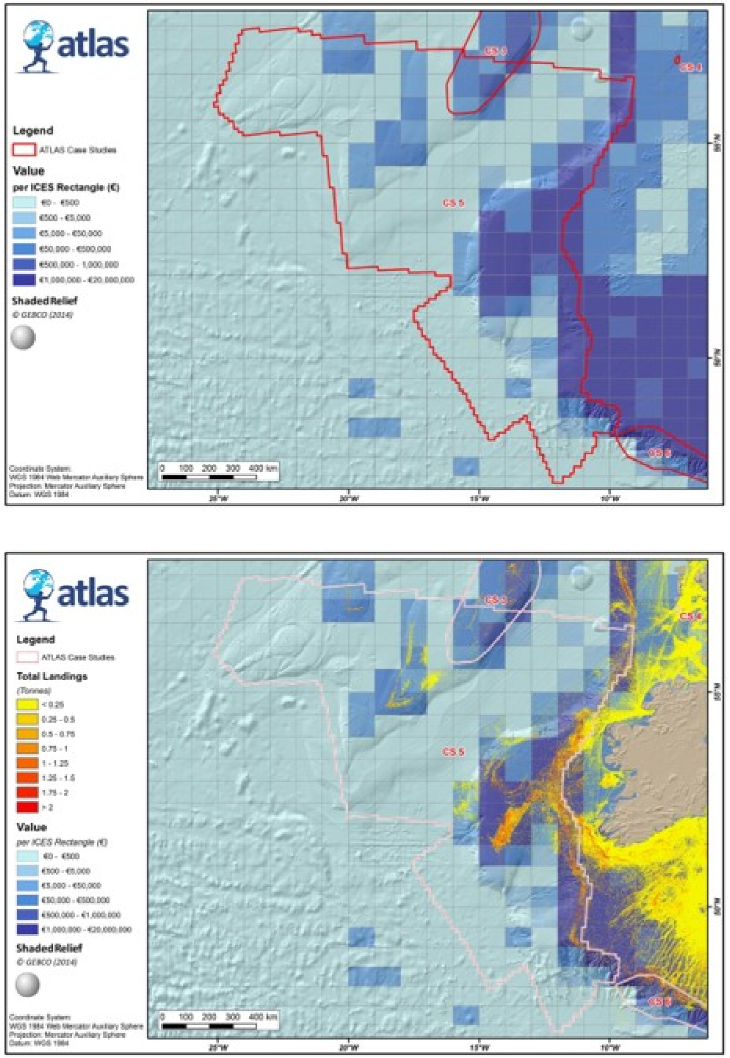



Figure 7: Value per ICES rectangle and deep sea demersal species landings in the Porcupine, Ireland

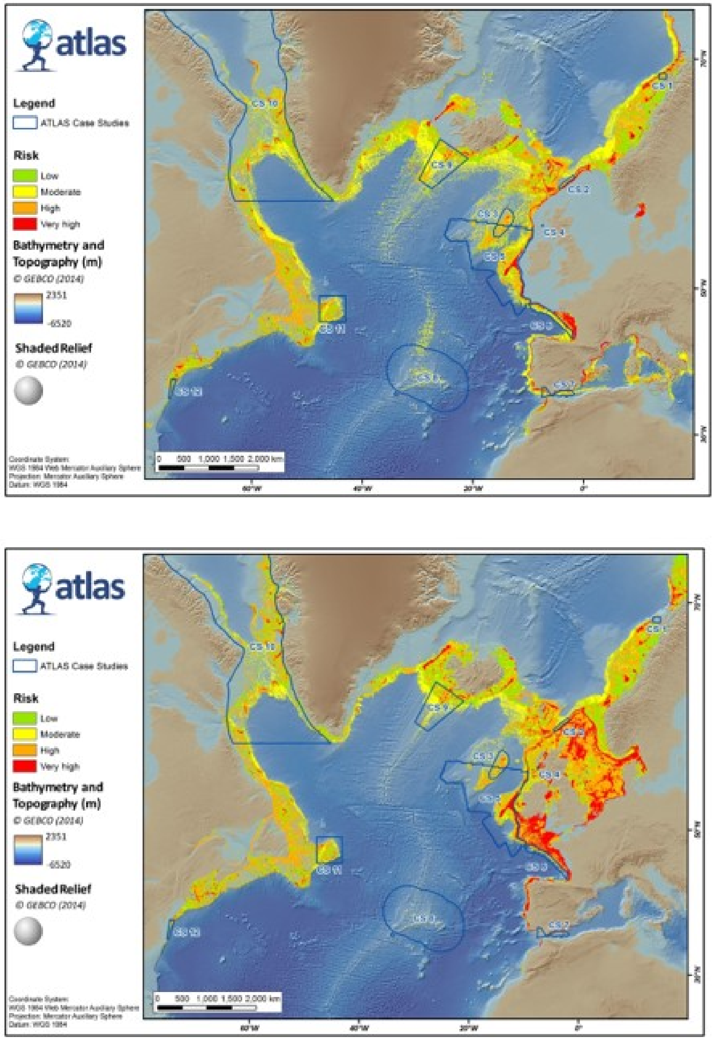



Figure 8: Assessment of risk posed to VME and fish habitat from pressures due to fishing activity across the North Atlantic Basin

### 3.4.4 ATLAS valuation surveys

Being able to value such a small proportion of services is not very satisfactory as a guide to MSP and decision making more generally. To help fill the gaps in valuation evidence, ATLAS is collecting empirical evidence on non-market deep-sea ecosystem services values for four case study areas: the Azores, LoVe Observatory, Mingulay (in the eastern Atlantic) and Flemish Cap (in the western Atlantic). These areas have been selected to cover the broadest range of both ecosystem types (seamounts, cold-water corals, sponge grounds) and jurisdictional regimes (national, European, ABNJ). National surveys for Mingulay in Scotland and LoVe in Norway have been completed and published (ATLAS Deliverable 5.4) while national surveys for the Azores and ABNJ for Flemish Cap (for Norway, Scotland and Canada) are in the final phase of data collection.

Results from Mingulay and LoVe suggest that the public in Scotland and in Norway show general eco-centric attitudes towards the marine environment, with broad recognition of the importance of ecosystem services, the current ecological crisis and the need for sustainable management, but only low to modest knowledge relating to deep-sea environments. The valuation experiment assessed trade-offs for improvement in a number of deep-sea environment attributes (environmental health and quality, increase in size of MPAs and new marine-related job creation). Results indicate that the public in both countries is willing to pay to support conservation of the unfamiliar deep-sea ecosystem, highlighting preferences for reducing specific marine pressures including marine litter and impacts on the health of fish stocks, followed by increasing MPA coverage generally, with least value ascribed to the creation of jobs. For example, for improving fish stocks to '>80% fish stocks healthy', the weighted average value for the Scottish public is €75 and for Norwegians is €179, while a change in deep sea marine litter densities from 'poor' to 'good', the Scottish public weighted average is €100 while for Norwegians it is €200.

Results such as these highlight that deep-sea ecosystems are seen as important by the general public, and provide support for further collective action required by the EU in moving beyond the 2020 MSFD objective of achieving good environmental status (in this case, in terms of Descriptors 3 and 10, commercial fish and litter) for Europe's seas.

Although at present the evidence base is incomplete, limiting the extent to which deep marine ecosystem services can be valued in monetary terms, methods and frameworks are available that can be put in place to support decision making with the best available information, including quantitative or qualitative spatial assessments of ecosystem services. Use of these frameworks will, in addition to directly supporting current decision processes, create additional demand for improved valuation evidence in the future. As the evidence based expands, there will be more opportunities to use economic valuation evidence to support deep marine and ABNJ management, monitoring and decision processes, and MSP in general.

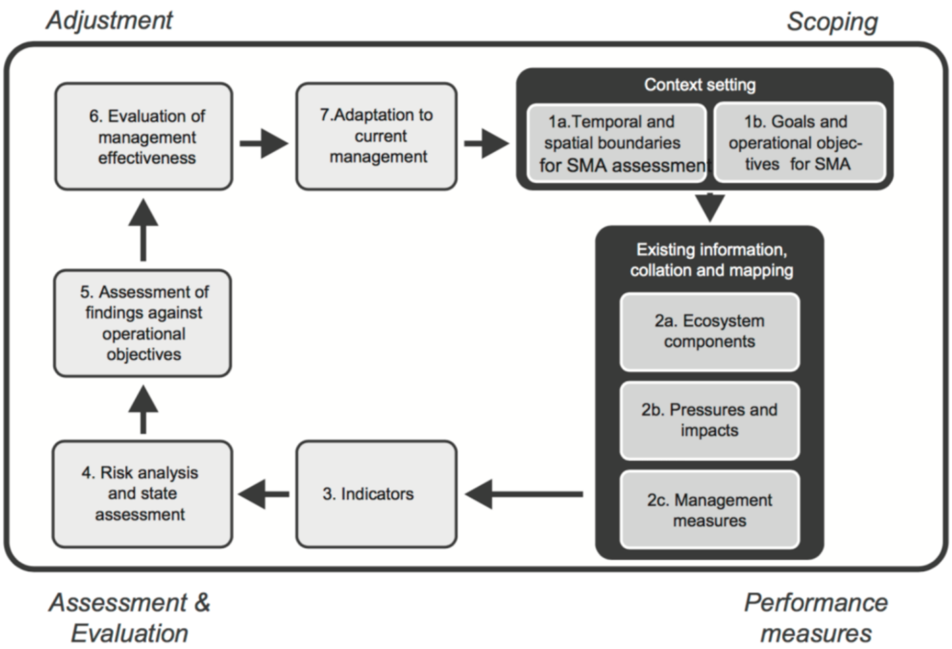
## 3.5 Valuation in MSP

The application of valuation in an MSP framework can take place in many ways: see Box 1 for possible entry points in the MESMA framework. At the most basic level, the following steps are necessary (Nordic Council of Ministers 2017):

- Select a suitable typology of ecosystem services to which activities can be linked
- Identify and define relevant marine/maritime activities and sectors .
- Assess how the environmental pressures arising from the activities impact on the ecosystem services

Economic valuation then seeks to assess changes in wellbeing for different stakeholder groups. Ideally, this is done through monetary indicators. To value deep-sea ecosystems and their goods and services, we need knowledge about the biodiversity, structure and functioning of the systems, and the factors influencing these, including the threats and pressures impacting on the systems, and how the systems and services respond over time (Armstrong et al 2010). Partly due to gaps in this knowledge base, little information of an economic nature is available, in particular for the deep seas, as explained above. Such information as does exist is mostly tied to the provisioning services of the ocean such as fisheries and fish habitat; with little information on regulating and cultural services, or future potential services from Blue Growth. Provisioning services such as fisheries are quantifiable, but regulating or cultural services are not well known at present. This lack of knowledge creates the risk of developing marine economic activities without sufficiently assessing the environmental impacts and trade-offs.

Hence, in many cases changes may have to be described quantitatively (physical units), semi-quantitatively (with scores), or qualitatively (text). Assessment may also go beyond simple cost benefit analysis (CBA) by including wider issues such as employment opportunities and distributional effects.




| Framework step | Think about possible roles for valuation evidence |
|---|---|
| **Step 1** definition of spatial and temporal boundaries, context, goals and objectives. | Some goals/objectives could be established in economic terms. Desires/needs of stakeholders, to consider here and throughout, could be assessed in monetary metrics, including estimation of public non-use values for natural systems and biodiversity. |
| **Step 2** collation and mapping of existing natural, economic and social information relevant to the objectives | Socio-economic components must be mapped and the (cumulative) impacts of these assessed. Values of activities/service could form one map layer. Tools such as InVEST, ARIES[1], MARXAN can facilitate this. |
| **Step 3** definition of indicators and related thresholds. | Values can be used as service indicators. Values as thresholds may be appropriate for subsistence uses or benefit sharing. |
| **Step 4** assessments of indicator states and risk analysis of management scenarios. | Economic assessment/comparison of scenarios, using CBA, MCA or portfolio analysis methods, including the tools identified above. |
| **Step 5** evaluating findings against the operational objectives. | In value terms where appropriate including assessment of trade-offs and sustainability. Possible role for ecosystem accounting. |
| **Step 6** assessing effectiveness of proposed management measures. | Effectiveness can be assessed in terms of realisation of potential values, and should include estimates of costs, opportunity costs, and cost-effectiveness. |
| **Step 7** collation of outputs and resulting management recommendations. | Including financing needs and potentially guidance on management instruments. Identification of valuation/data gaps and proposals to address them. |

*Box 1. Steps in the MESMA Framework and possible associated uses of valuation evidence.*

As discussed above, supporting services are commonly omitted from valuations and appraisals in order to avoid double counting. However this is only appropriate if the boundaries of the assessment encompass all the final services. In the case of the deep sea, it will often be the case that the spatial boundaries do not in fact do this, because the final services supported arise elsewhere. In this situation, MSP should recognise (and where possible and appropriate, value) supporting services.

A further complication is that services may not be spatially fixed – e.g. fish stocks – creating a challenge for spatial valuation in MSP. This can be partly addressed through setting appropriate scales and boundaries, and cooperation across governance scales to make robust assessments of trade-offs (European Marine Board 2019). Global change will also impact on ecosystems, services and values in ways that may be both complex and uncertain, both directly through changes in water temperature and chemistry, and through impacts on connectivity, as discussed above.

This makes total valuation a demanding exercise, and most marine-related valuation studies focus on coastal environments. However, there have been some studies carried out on the deep sea and in ATLAS case study countries including Ireland, UK and Azores, for example for cold water corals, as discussed above, and the ATLAS project itself is further enrichening the evidence base via direct valuation studies in four case study areas.

# 4 Barriers and opportunities

Recently, the European Marine Board Future Science Brief (EMB 2019) showcased current thinking in ecosystem service valuation for the marine environment. They reported that, although ecosystem valuation has advanced significantly over the past decade, results are rarely used to support marine management and policy. They conclude that making ecosystem valuation an integral part of marine management decision models would both improve decision making and enhance the evidence base.

Similarly, Adams et al. (2019) describe various ways of considering the disconnect between the accumulation of evidence on ecosystem service and values and the use of this evidence in conservation planning, referring to the “research-implementation” gap, the “assessment-implementation” gap, the "knowing-doing" gap and the “implementation crisis” (Pfeffer and Sutton 1999, Knight et al 2006, Knight et al. 2008, Knight et al 2011). They stress the need to facilitate the transition from assessment to implementation: not just accumulating knowledge, but explaining where, when and how it can be used to support decision making.

This problem of a disconnect between the potential of monetary valuation and its actual impact in supporting policy, decision-making and day-to-day management is not specific to the marine environment, but may be particularly stark in marine, and especially deep-sea, settings, for the reasons outlined above, including the relatively thin evidence base, the remoteness of and lack of familiarity with these systems, and the spatial and temporal disconnect between ecosystem function and final service provision. It is a complex problem, with roots in numerous concerns and controversies regarding the framework and tools of environmental valuation, some of which are justified, some less so.

## 4.1 Controversies regarding valuation

Using market values to account for goods and services traded in markets, including ecosystem goods (such as fish) or non-biotic resources accessed through ecosystems (deep sea minerals, oil and gas), is relatively uncontroversial. But use of economic values for non-marketed services such as climate regulation or biodiversity protection has been criticised on many fronts (Table 2).

*Table 2: Valuation assumptions, problems and resolutions*

| Assumption | Problem? | Generalisation | Resolution? |
|---|---|---|---|
| Individuals' preferences strongly correlated with welfare | Sometimes false (e.g. drug addiction), often dubious (e.g. myopic preferences and regret: Hoch and Loewenstein, 1991). | Democratic societies allow wide freedom of choice under rules to curb excesses, encourage saving etc. | Recognise TEV focuses on individual preference, consider other moral decision rules in deliberative processes. |
| Individuals have information and ability to have stable, well-formed preferences they express through decisions | People have "bounded rationality" (March and Simon, 1958), construct preferences (Slovic, 1995), especially for hypothetical decisions, unfamiliar goods/services | Affects other methods. Market institutions consistent with assumptions, with limits (advertising, trade descriptions…). | Cognitive limits may support *procedural* rationality (Laville, 2000). Reduce bias via information, thinking time, deliberation. |
| Interpersonal comparability of utility | Identical indicators of benefit to different individuals may represent different levels of human welfare | Affects any system (including voting systems), not limited to monetary units. | Practical option is to act 'as if' comparisons reliable, and use income weighting. |

| | | | |
|---|---|---|---|
| | (d'Aspremont and Gevers, 2002). | | |
| Values constrained by ability to pay | Raw value estimates assume that income distributions are desirable/fair | Tax/benefit policies redistribute incomes so actual distributions partly reflect democratic processes. | Income weighting to adjust values in transfer/appraisal. |
| Smooth, continuous value functions | Non-linearities, threshold effects and areas of highly inelastic demand / rapidly changing values | Small-scale, marginal assessments less likely to suffer than large-scale, major changes. | Valuation less useful for critical natural capital or potentially catastrophic changes. |
| Data gaps in scientific understanding and valuation evidence | No valuation or appraisal can be complete and accurate | Applies to all methods: use range of values, sensitivity analysis, clear statements of gaps. | Valuation/appraisal are aids to deliberation, not “the answer”. |
| Optimism bias: tendency to underestimate future costs and overestimate benefits | CBA likely to be biased (see Mackie and Preston, 1998). | More about physical outcomes and timings than valuation methods. | Recognise and adjust for optimism (or ‘pessimism’) bias. |

Economists recognize all these issues, but use TEV and CBA for practical reasons: on the one hand, many of the objections can be adjusted for to some extent, while on the other, no alternative approach is perfect. The key issue is whether or not the evidence is actually useful, in terms of improving decision-making. There are widely divergent views on this question: Flyvbjerg (2009) argues that errors in forecasting are so substantial that CBA will almost always be “strongly misleading,” summarising this as “Garbage in, garbage out”; Asplund and Eliasson (2016), in contrast, conclude that, despite the pervasive uncertainties, CBA “is able to fairly consistently separate the wheat from the chaff” and thereby contribute to substantially improved decision processes and outcomes. Of course which is closer to the mark will depend not only on the characteristics of the decision to be made and the ecosystem/situation concerned, but also on the deliberation and decision-making processes used, and (crucially) the individuals involved, and their knowledge, skills and perspectives. In a review of CBA of conservation projects, eftec (2010) found that, while there are very few clear examples of “near-perfect” CBA studies, there are several examples that are “good enough” to provide a useful aid to decision making within a given context.

Nevertheless, these issues and biases lead to understandable concerns regarding the validity and reliability of valuation evidence as a guide to thinking and decision support for marine environments. Of course it should be recognised that there are different purposes and uses for valuation evidence, including spatial planning, but also awareness raising, ecosystem accounting, appraisal for specific policies, projects or siting decisions, demonstrating value for money, calculating compensation for environmental damages, and so on. Each of these may call for different specific methods and coverage, and different requirements for accuracy and research expenditure commensurate with the context (Barton, 2007). Valuation could therefore be rejected for one purpose, but still found useful for another.

For example, a focus on trade-offs, comparisons of "states of the world", and what may be lost or gained from decisions is more policy relevant than estimates of the "absolute" value of ecosystems or their services, which make for catchy headlines but "have no specific decision-making context" (Costanza *et al*., 2014). Taking account of relationships and feedbacks at broad scales – as in MSP – can help to defuse the objection that multiple projects change prices and substitute sets in ways that conventional appraisals overlook (Hoehn and Randall, 1989). Increasingly, attention is turning also to environmental and ecosystem accounting, calling for different types of value (exchange values rather than TEV, as explained above), and many policy assessments consider economic impacts (contributions to gross value added and employment) as well as, or instead of, welfare-based estimates.

The value of a good or service can vary with its quantity and quality and hence most values represent a marginal value relating to a specific context. For many environmental services, demand can be quite 'elastic' at high levels of provision, but inelastic for lower levels, and effectively 'infinite' for essential services or 'critical natural capital' (Chiesura and De Groot, 2003; Figure 9). This does in effect put limits on the applicability of valuation: valuation is relatively unproblematic under elastic demand, less reliable under inelastic demand, and not appropriate for critical natural capital (Farley 2008). In the context of MSP, this suggests that valuation could be a useful tool for informing trade-offs and reducing opportunity costs, but only within limits set by the overarching need to ensure that the sustainability objectives are met.

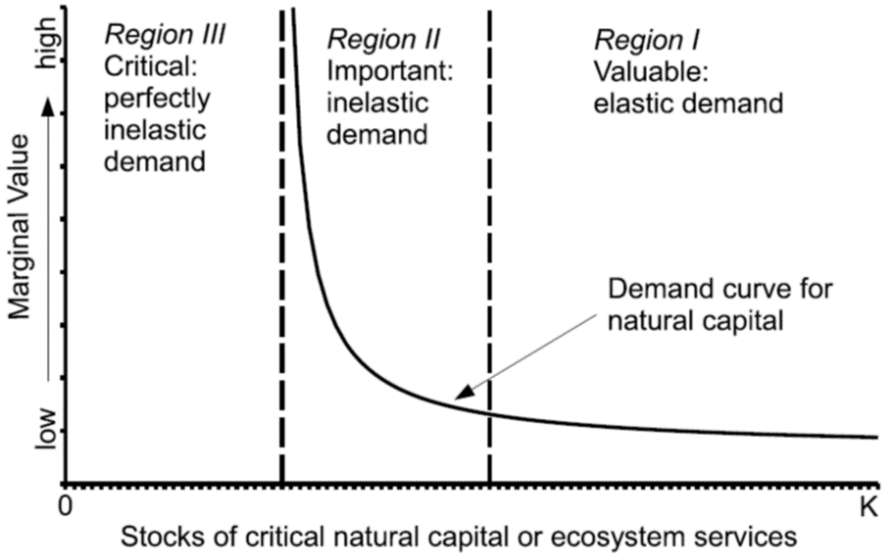



*Figure 9: Demand for natural capital and implications for valuation (Source: Farley 2008)*

Coverage in any case is limited to the estimated part of total economic value. In practice this rarely covers all sources of value, due to incompleteness of the evidence base linking environmental features to valuable services, and in some cases to conservative assumptions designed to avoid double-counting, or reluctance to use estimates of non-use values that are seen as less reliable or less credible. This is a frequent issue in cost-benefit studies: there is often a concern that the stated preference (SP) surveys used to assess non-use values may be detecting some part of use values too, and that including both the SP results and the ecosystem service values derived by other means could result in some double counting. Some studies present arguments regarding which other services are thought to be covered and which not. Luisetti (2008) for example uses stated preference for "composite environmental benefit" intended to cover wide range of impacts (recreation, aesthetics, water quality, biodiversity) but includes separate values for fisheries benefits and climate regulation, which are thought not to be considered by respondents in formulating their SP responses). Other studies (such as the UK marine conservation example discussed below) avoid adding the numbers together but hold the non-use values "in reserve" as a further argument for the robustness of positive benefit-cost ratios (BCRs) – i.e. arguing the case for conservation based only on use values, while pointing out that additional values will exist.

Though it is clear that the default zero values in such cases are underestimates, this can be preferred from a tactical perspective in that it enhances the credibility of the value components that are estimated and avoids 'contaminating' them with the uncertainty of the less accurately estimable components, while at the same time leaving space to stress the importance of the additional elements that are not 'counted' in the value estimate

Nevertheless, critical elements of the natural environment may be overlooked in decision processes if they are not recognised as important. For example, we may be unaware of all the ways deep sea ecosystems support services and human welfare: but this is a problem for decision support generally, not just valuation. Primary research can help, but in terms of the pressing need to develop MSPs, it is not practical to wait until scientific uncertainty is resolved. MSPs need to be formed under uncertainty, making best use of the information available. Consequently, results of decision support tools used in MSP need to be tested for sensitivity to assumptions used, and any risks, uncertainties, missing data, and other caveats must be clearly and fully reported.

### 4.1.1 Example: deep sea mining

Valuation could be particularly relevant for deep sea mining, for example, where the need to demonstrate limited and acceptable levels of environmental impacts could be addressed in part via valuation studies. Deep Sea Mining (DSM) is the extraction of metal resources from the deep sea bed, a potentially rich source of key minerals. Some deposits lie beneath national waters and are sovereign resources. Many others lie in the Area Beyond National Jurisdiction (ABNJ, 'the Area') and access to these resources is regulated by the International Seabed Authority (ISA) under the United Nations Convention on the Law of the Sea (UNCLOS) (Baker and Beaudoin, 2013).

There is potential for these deposits to be highly profitable, and, if shared fairly, to meet strategic needs for resources as well as fuelling social and economic benefits for less wealthy nations. Deep sea deposits are generally richer in mineral content than terrestrial sources, and are on or near the seabed surface, and so can in principle be accessed with creation of much less spoil than terrestrial mines. Together with their location far from human populations, this might make DSM environmentally preferable to terrestrial mining. On the other hand, DSM activity would inevitably impact on deep sea ecosystems with effects that are extremely uncertain and may be long term or irreversible. At the same time, the economic and social impacts of DSM could be highly variable depending on the mechanisms put in place to regulate and tax the activities.

Therefore, governments and the ISA face difficult choices regarding the appropriate development and regulation of DSM. The decisions to be taken include broad-scale/strategic decisions about the global approach to licensing exploration, standards and protocols for seeking operational permits, research and monitoring requirements, and benefit sharing. They also include more localised decisions about specific resource deposits and applications to exploit them.

To make good decisions, it is important to understand the potential economic, social, and environmental impacts, pros and cons of DSM. This also requires a broad strategic view of DSM and its potential role in achieving sustainable development, meaning that DSM should not be considered in isolation, but as a part of a world economic system that includes terrestrial mining and other alternatives to mining such as recycling and resource substitution. It is also essential to recognise the full range of values arising from the deep sea, including all the ecosystem services discussed above.

The issues of whether and how – and when – to mine the deep seas is divisive because DSM will have impacts on all aspects of sustainable development: economic, social and environmental. This includes potential long-term and/or irreversible environmental impacts, about which there are substantial uncertainties and unknowns. There are also unresolved questions regarding sharing the benefits and costs of DSM, and the long-term social and economic consequences for resource-owning societies.

Difficult decisions will have to be made, and probably sooner rather than later, because commercial pressures within the deep sea mining/technology industry, sustainable development concerns for resource owning states, and broader strategic and economic interests at a global scale, all represent drivers for pushing forwards with DSM. Especially given the substantial uncertainties and unknowns stressed above, it is therefore important that we consider what our choices are, what policy options we have for implementing them, and what methods we have for evaluating and informing decisions through appraisal and valuation of the impacts, in order to ensure that DSM is fully incorporated within the MSP process.

Wakefield and Myers (2016) attempt to use cost-benefit analysis based on 'realistic yet hypothetical' mining scenarios developed for three mineral deposits in the Pacific Island Region. They argue that the results 'indicate that deep-sea mining has the potential to increase the well-being of the people' in two of three case studies. However, while their initial profitability analysis is highly relevant to determining situations under which DSM might be considered, the assumptions and analogies adopted in the absence of data on the impacts and values mean that CBA is of very little use for the purpose of deciding whether or not it should go ahead. This is particularly the case since the pervasive uncertainties mean that the key issue is not calculating the expected benefit of each option, but rather developing an appropriate strategy following the precautionary principle, including consideration of both the potentially high value of methodical research and monitoring, and the highly asymmetric nature of potential regret from decisions to mine now (irreversible) or to delay until information is better (reversible).

### 4.1.2 The role of Area-Based Management Tools

A key element of MSP is to achieve spatially explicit management, including zoning of different activities and pressures, including through the use of Area-Based Management Tools. One such tool that is particularly important for achieving sustainable management of marine environments is the creation of marine protected areas (MPAs). There is a global movement to implement MPAs (Pelletier 2018) and the World Database on Protected Areas[23] shows a rapid growth in MPAs from approximately 0.7% of the ocean in 2010 to 7.4% in 2018, with a total area of almost 27 million km$^2$ protected to some extent. Nevertheless, only 2% is in implemented strongly or fully protected areas. (Sala et al 2018). The United Nations' target for global ocean protection is 10% of the ocean in Marine Protected Areas (MPAs) by 2020.[24]

The headline figures mask a large divergence in protection depending on the governance status of the waters: while 17.3% of national waters are protected, only 1.18% of the ABNJ is covered[25] and effective protection of these areas remains a particular challenge (Johnson et al 2018). Facing up to this gap, the UN General Assembly agreed[26] in 2015 to develop an international legally binding instrument under UNCLOS on the conservation and sustainable use of marine biological diversity of areas beyond national jurisdiction (BBNJ). Negotiations for the instrument cover ABMTs and MPAs.

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[23] https://www.iucn.org/theme/protected-areas/our-work/world-database-protected-areas

[24] Aichi Target 11 : https://www.cbd.int/sp/targets/

[25] https://www.protectedplanet.net/marine#distribution 11 January 2019

[26] UNGA Resolution 69/292 : http://www.un.org/en/ga/search/view_doc.asp?symbol=A/RES/69/292

For the ABNJ of the North Atlantic, ATLAS research (Johnson et al 2018) has identified over 50 ABMTs and other areas upon which future ABMTs could be based, considering only OSPAR MPAs, CBD EBSAs, and areas closed by North Atlantic RFMOs to protect VMEs.

A second example is in the designation of MPAs. These can be effective tools for ecosystem protection, in particular if they consider basic ecological principles and set clear conservation goals (Saarman et al 2013). While MPA benefits generally increase with size (Edgar et al 2014) large MPAs can be difficult to implement for social or political reasons (IUCN 2008, McLeod et al 2009) Consequently, efforts for designing effective MPAs have shifted focus from establishing individual MPAs to networks of MPAs (MPAn) (Smith & Metaxas 2018)

The Azores report (UNEP/CBD 2007) sets out criteria for identifying ecologically or biologically significant marine areas and designing representative networks of MPAs. These are:

| **Criteria for ecological/biological significance** | **Criteria for representative networks** |
|---|---|
| Uniqueness or rarity<br>Special importance for life-history stages of species<br>Importance for threatened, endangered or declining species and/or habitat<br>Vulnerability, fragility, sensitivity, or slow recovery<br>Biological productivity<br>Biological diversity<br>Naturalness | Ecologically and biologically significance (see left column) and also:<br>Representativity<br>Connectivity<br>Replicated ecological features<br>Adequate and viable sites |

Thus there are several principles beyond the site level for configuring MPA networks. However these relate to the ecological desiderata. In practice, tools are also needed for estimating and taking account of the economic aspects of protection - and of failure to protect – such that MPAs can play their full role within the wider framework of MSP. These include both estimates of the economic opportunity costs of MPAs – the growth and jobs foregone from not exploiting resources in the area – and the benefits, including market benefits associated with enhance fisheries productivity offsite, and the non-market TEV values of ecosystem services protected and enhanced by the MPAs.

In this context, there is a growing literature on the costs and benefits of MPAs, however relatively little of this examines both costs and benefits together, or applies a full cost-benefit approach. There are cost estimates (Balmford et al 2004, Sumaila et al 2007) and studies of marine reserve benefits (Russ et al. 2004, Gell and Roberts 2003, Halpern 2003). But there is little that combines monetary estimates of costs and benefits. Work on cost-effectiveness is more common, in particular in the context of achieving a given area or standard of protection at least cost (Wainger et al 2010).

Some studies that adopt a cost-benefit framework nevertheless do not qualify as cost-benefit studies due to a lack of data preventing valuation of key parts of the appraisal. For example the Sumaila et al (2007) study uses a cost-benefit framework to assess “Potential costs and benefits of marine reserves in the high seas”, but the only monetary estimate is of the opportunity cost of lost fish production in the short term. Longer term benefits, including fishery gains and reduced risks, are discussed but not quantified. The paper nonetheless presents a strong argument for some increase in protection of the high seas: the estimated opportunity costs are only US$270 million annual profit loss from a 20% closure of all pelagic and deep sea fisheries, and it is noted that about US$152 million per annum is currently paid as subsidies to high seas deep-sea bottom trawlers alone.

This is a good example of the usefulness of the cost-benefit framework as a rational and methodical approach to structuring and presenting information for constructing arguments and decision support: even if it is not possible to put monetary figures on all or most of the impacts, the results of the exercise can still be highly policy-relevant, and informative for MSP purposes.

One example of more comprehensive cost-benefit analysis is the study of the UK Marine and Coastal Access Bill’s provisions for Marine Conservation Zones, as set out in the Impact Assessment (Defra 2009) and the documents supporting that (McVittie et al 2008, Moran et al 2008; Hussain et al 2010; ABPMer 2007). The analysis is applied at national scale, as there is little evidence at the individual site level. The study identified eleven ecosystem service impacts and attempted to value seven of these:

- food based on market values
- raw materials based on market values;
- recreation on expenditure;
- nutrient cycling on the benefit transfer from Costanza et al. (1997);
- climate regulation on primary productivity and the official UK public sector carbon value;
- coastal defence on avoided damage cost;
- “cognitive values” on the value added from research spending and expenditure on education with specific marine focus.

A separate stated preference (SP) survey was carried out for non-use values, but these are not treated as additional in order to avoid possible double counting.

The study concluded that the establishment of a network of MCZs throughout UK waters has a positive BCR of between 6.7 and 38.9. Although this is an imprecise conclusion based on far from perfect evidence about benefits, the results are reasonably robust in the sense that sensitivity testing shows that even given the uncertainty in the estimates it is rather unlikely that the BCR could be below 1, and this is *a fortiori* the case given that the non-use benefits are not included in the BCR calculation.

The study is a good illustration of the use of expert judgement to score likely impacts where we have some evidence of the total value of a service, but limited evidence of the impact on that service of a specific policy change. This kind of uncertainty is quite pervasive in studies of conservation decisions, and there are different approaches to it. Some studies in effect push the scientific uncertainty into the valuation study, using stated preference studies of willingness-to-pay for conservation actions or results without actually modelling the ecological relationships. More recently, there has been a greater focus on use of one or other ecosystem services framework, explicitly breaking impacts down to individual services and attempting to value them separately. This puts greater emphasis on issues of missing data, and the use of expert judgement is one way of trying to deal with this – for example, using Delphi surveys as in the ATLAS project.

Intuitively it makes sense that we might expect more accuracy from letting experts make the judgements on scientific and ecological relationships, and limiting valuation tasks to clearly specified outcomes. However, where stated preference is used, this does depend on respondents being able to think of different impacts separately. If in fact there are strong linkages between impacts – for example, conservation of a particular fish species might not be possible without conservation of spawning areas, settlement areas and feeding habitats and good environmental quality in all of them – then it may not be reasonable to expect respondents to overlook these linkages, and stated preference valuation is likely to involve valuation of the conjoined changes. Where this is the case, even though the assessment framework breaks impacts down into all the component ecosystem services, it may still be preferable at the valuation stage to use composite values that cover several service categories, and hence avoid double counting.

A very different approach is typified by the Homarus Ltd (2007) study of a proposed conservation zone centred on Lyme Regis, UK. A statutory two hundred and six square kilometre 'closed area' in Lyme

Bay, South West England entered into force on the 11 July 2008 to protect the reef substrate and the associated biodiversity from the impacts of trawling and dredging with heavy demersal fishing gear. Within this area, scallop dredging is stopped, but more sustainable forms of fishing are allowed (e.g. dive catching of scallops, crustacean potting and fixed netting of skates and rays), as is recreational use. The Homarus research uses a partial cost-benefit approach, focusing only on the market returns from different options. Since the market returns from the protection option exceed those from the business-as-usual case, this provides good evidence that protection would be beneficial, given that the environmental benefits of protection are unknown but certainly positive.

This strategy is quite common, whether in the starkest form of focusing only on the market returns, or in milder manifestations in which certain more easily quantifiable ecosystem service impacts (for example carbon sequestration) are included. The basic argument is that if the economic impacts plus easily valued services are themselves enough to justify a project, and in addition there are other ecosystem service benefits that cannot be valued, but are unequivocally positive and therefore can only strengthen the result, that's sufficient for informing policy. This argument is fine so far as it goes, but of course only applies in the most "obvious" decisions where the upside is strong and the downside largely absent.

In the Lyme Bay case, the Defra (2008) Impact Assessment concluded that the Homarus report did improve the understanding of the relative importance of all activities in the closed area, but that it underestimated the value of the MPA to the scallop fleet (through the assumption that it represented 11.3% of catches in the two adjacent ICES rectangles). Curtis and Anderson (2008) went beyond direct costs to the fishing sector to assess the wider social and economic impact of the MPA on the fishing industry; but that was considered an overestimate as the MPA was assumed to represent between 25 and 50% of the landings from the two adjacent ICES rectangles. Hence the impact assessment stressed the 'limitations and caveats' around the specific figures, but considered them to give a useful indication of the scale of the costs to be weighed against the wider economic, environmental and social benefits, and hence a valuable input to deliberation and decision making.

Rees et al (2010) argue, in contrast, that since the economic reports present very different outcomes that can be traced back to different assumptions applied to the same area and data available, "the Lyme Bay case study illustrates that reliance on market valuations and resource use decisions based on traditional neo-classical economics can obscure other issues pertinent to the ecosystem approach concerning whether ecological features should be protected." This is despite the fact that the Defra Impact Assessment shows transparency as to how these figures were attained: they suggest that adopting an ecosystem approach within marine spatial planning can reveal improved ("win-win") solutions for the long-term based on a thorough evaluation of the environmental, social and economic values of marine biodiversity.

The ecosystem approach demands that environmental, economic and social sustainability are balanced in the decision-making process (Laffoley et al 2004). The process of making choices as an individual or as a society about ecosystems and their use implies a process of valuation (monetary or non-monetary) of the respective parts (Costanza et al 1997). Conflict arises between stakeholders as the concept of value is broad and multifaceted, including social, monetary, emotional, environmental and/or cultural aspects. A win–win situation demands that all these aspects of value are understood, and stakeholders agree upon an equitable balance of benefits from resource use. This does not, however, rule out the use of estimates of economic costs and benefits based only on assumptions about fisheries, and it may even be that these are the only monetary figures that can be derived in a reliable fashion in some marine environments. It does, on the other hand, suggest that decisions should not be taken based on those estimates alone, and that some means of expressing and taking account of the wider costs and benefits must be found – as in the ATLAS assessments presented in section 3.4 (*Figure 5*, *Figure 6*).

## 4.2 Role of decision makers

The availability of more or less reliable evidence regarding ecosystem service provision and values is only half of the story with respect to the potential of values to inform policy and decision making. The evidence has to be taken up and used by decision makers or other stakeholders in order to have any impact.

In this context, the views of different stakeholders regarding the validity and legitimacy of values are important. This includes views regarding the legitimacy of the valuation framework overall – i.e., is economic value an appropriate way to think about decisions regarding the marine environment? – as well as on the reliability of different methods for measuring values. These issues are explored in more detail in ATLAS Deliverable 5.3.

While monetary valuation has been controversial, this can be interpreted in the context of gradual progression in the framings of human-environment interactions. Although many stakeholders may find the idea of "pricing" the environment somehow distasteful or unethical, on a practical level, Mace (2014) recognises that most environmental decisions are made on the basis of economic arguments. She argues convincingly that the position of refusing to engage with valuation risks further marginalisation of nature from decision-making: "If the benefits provided by nature are assigned no value, they are treated as having no value, and current trends in the decline and deterioration of natural systems will continue." At the same time, strongly reductionist approaches to valuation are set in a 'nature for people' framing that is most likely to elicit rejection on principle. A softer 'people and nature' framing is more acceptable, and it is towards this that many initiatives (such as IPBES) are tending, although this may represent a challenge for existing valuation methods.

Although valuation is primarily used in public sector settings, there is increasing uptake in the private sector, for example for determining customer priorities, assessing impacts and dependencies on natural systems, communication and performance tracking including natural capital and ecosystem accounting. Initiatives at the international scale include, for example, The Natural Capital Project[27], the World Business Council on Sustainable Development[28], and the Natural Capital Coalition[29].

Rose et al (2017) use the concept of 'policy windows' (Kingdon, 2003), periods that are particularly opportune for scientific knowledge to be incorporated into policy, due to events in 'process streams': a problem becoming more obvious or salient; a new policy opportunity arising; changes in political forces or agendas. For example, Rose et al (2018) illustrate how such a window was effectively utilised in the case of the Lawton Review (Lawton et al, 2010). Moon et al. (2014) argue that such opportunities can arise suddenly and disappear quickly, while Rose et al (2017) argue that it is important to foresee (or seek to create) these chances, to build the capacity to respond quickly and

[27] www.naturalcapitalproject.stanford.edu

[28] www.wbcsd.org

[29] www.naturalcapitalcoalition.org

effectively with arguments framed for the audiences, and to persevere with arguments and push incremental progress.

The current dynamic towards greater use of monetary valuation in general, in both public and private sectors, together with the need to generate Marine Spatial Plans, represents such a window of opportunity to enhance the use and effectiveness of valuation evidence in the management of the marine environment in general, and the deep seas in particular.

Overcoming the disconnect between evidence availability and use in policy will require making the tools and evidence more familiar and acceptable to decision makers. The literature on motivated reasoning suggests that individuals presented with information that is not consistent with their existing knowledge and values are likely either to fail to assimilate the information or to interpret it in a way that strengthens rather than dispels misconceptions (Hart and Nisbet 2012), suggesting a need to contextualise knowledge in ways that enable audiences to understand the relevance in the context of their problems, decisions, and values. Science-policy interaction mechanisms based on linear knowledge transfer – knowledge' speaking 'truth' to 'power' (Wynne et al. 2007) – often fail to influence policy makers' or public behaviour (van Kerkhoff & Lebel 2006). Sutherland and Wordley (2017) characterise this failure as "evidence complacency" and discuss a number of possible causes.

However, Evans et al (2017) argue that framing the problem as one of 'complacency' is likely to be counterproductive, preferring to recognise that the incorporation of evidence in decision processes "is slow, non-linear, inherently political, and based on relationships and links between multiple societal actors with a stake in a particular issue" (p1588). When uncertainty is high, and values are contested, science-policy interactions are more influential when they facilitate multi-way interaction processes between science, policy and stakeholders that contribute to real changes in the understanding and behaviour of policy makers and other target audiences (see Pielke 2007). This requires iterative processes of dialogue to enhance the credibility, relevance, and legitimacy of communication (Sarkki et al 2015), as well as strategies for framing arguments in ways that match the interests of audiences. Cognitive dissonance created as part of dialogue in a supportive learning environment can be a valuable method of stimulating creative thinking and problem solving, in contrast to the defensive reactions triggered by dissonance in a non-consensual or threatening framing (Fischer et al. 2011).

Addressing some of the barriers identified above can be carried out through a combination of research to enhance the evidence base and work to make stakeholders in general, and decision-makers in particular, more familiar with valuation concepts and evidence, and the strengths and weaknesses of the decision support tools available.

EMB (2019) make a number of specific recommendations to improve the situation, including the need to “Include ecosystem valuation in marine management decision models”, “Develop the Natural Capital Approach and Natural Capital Accounting”, and “Create open databases that contain the data, meta-data, applied methodology and results of marine ecosystem valuation studies (monetary as well as non-monetary).”

It is widely recognised by economists (see e.g. TEEB, 2010, Diaz et al., 2018a,b) that monetary valuation and cost benefit analysis only provide one form of evidence to support decision making, that should be used as a complement to other forms of ethical, social, economic and natural science analyses, and consideration of various opinions relating to environmental exploitation and conservation. Monetary valuation, and decision support using it, should never be treated as the ‘right’ answer, nor as an alternative to wider deliberation. Rather, valuation is a support to thinking about difficult decisions, and a way of summarising certain forms of information in a convenient and tractable fashion. Of course there remains a risk of results being misused – for example being “cherry-picked” to support pre-determined conclusions – but this problem is hardly unique to valuation.

Other techniques exist that elicit different expressions of social preferences, including deliberative monetary valuation, ranking, participatory multi-criteria analysis, citizen juries, in-depth discussion groups, participatory modelling and mapping, and so on. These alternative decision support and/or valuation methods do not resolve all the concerns identified for valuation and CBA, and may introduce new ones, but can be useful in allowing different perspectives on social choice. Alternative decision rules may for example prioritise precaution and robustness over maximisation of expected values. In many cases these methods can be complementary to valuation and CBA, with evidence from several methods being incorporated within a wider deliberative process.

In all events, it should be clear that valuation is one aspect of decision support. Values, and tools such as CBA, are useful as aids to structuring diverse information relating to complex processes and decisions, and informing deliberation and reflection on the management options and their consequences. They should never be treated as “black-box” methods for giving “the answer” to a particular problem. But at the same time, they have huge potential for informing deliberation in environmental decision making in general, including in Marine Spatial Planning. The opportunity to extend the use of valuation evidence in MSP, where this is feasible and appropriate, should be seized, in order to strengthen the deliberative processes and improve outcomes.

# 5 Conclusions

Although our understanding of the functions and processes in marine ecosystems, their reactions to human pressures, and their contributions to human economic activity and well-being remain limited, it is evident both that marine ecosystems represent vital and valuable assets for humanity, and that they are at risk from a range of anthropogenic pressures (Luisetti et al 2014). Developing better tools for their sustainable and adaptive management is a priority to which monetary valuation techniques have the potential to contribute.

Monetary valuation is one manifestation of a model of how aspects of the natural world influence human wellbeing. Like any model, the important issue is not whether it is ‘right’ or ‘true’, but rather whether it is ‘useful’. Thinking about it in terms of attempting to represent an underlying truth is not helpful, and makes for rather an easy straw man. It is much more interesting to consider whether or not valuation is useful as a decision support tool, both in particular localised circumstances, and as an aid to general strategy formation and large-scale, dynamic environmental planning and management. At a minimum, the valuation framework provides useful ways for thinking about how and why humans might value aspects of nature. This is not limited to selfish/’capitalistic’ concepts of nature, and in particular the total economic value framework does provide explicit space for non-selfish preferences (non-use – existence, altruistic, bequest values) and also for uncertain uses (insurance values). Similarly, the ecosystem services framework provides a useful checklist of ways in which natural systems provide benefits to humans. Of course, there is no claim that these values and benefits are an exhaustive representation of natural values, merely a minimum set of things to consider, alongside wider cultural, social and ethical perspectives.

Naturally, the estimation and uses of economic values for services such as biodiversity protection or climate regulation can be contentious, and this creates a certain reluctance to use valuation evidence in some quarters. And valuation is not essential: there are alternative ways of carrying out appraisal and policy planning: for example, using multi-criteria approaches or collective decision methods. However, valuation may have a number of advantages, including making the processes easier, more defensible, more transparent, and/or more (cost-) effective. In particular, arguments for recognising the importance of remote marine ecosystem services might be more convincing, for some decision makers or contexts, when they are expressed in monetary values that can be compared with the values of marketed marine products.

The majority of existing ABMTs are likely to become less fit for purpose or redundant within the next 20–50 years (Johnson et al 2018). Alongside the current need to develop Marine Spatial Planning at national and oceanic scales, there is a clear opportunity to increase the use and usefulness of valuation evidence. Optimisation software has been applied to try to identify the best spatial design for protecting multi-species connectivity (D'Aloia et al 2017), but four fundamental sources of uncertainty – process, measurement, model, and causal – must be considered. (Carr et al 2010). To evaluate priorities for ABMTs, higher resolution, smaller scale predictions for the next two-to-five decades are needed (Johnson et al 2018). The two main problems are data availability and model methodology, and the asynchrony between data/knowledge acquisition and pace of change is particularly worrying. This applies across all areas of evidence – physical, biochemical, ecological and economic – and ATLAS is contributing to enhancing the baseline evidence available for planning in all of these domains.

## References

Aanesen, M., C. Armstrong, et al. (2015). "Willingness to pay for unfamiliar public goods: Preserving cold-water coral in Norway." Ecological Economics 112: 53-67.

ABPMer, Risk and Policy Analysts, & Jan Brooke Environmental Consultant Ltd. 2007, Cost Impact of Marine Biodiversity *Policies on Business - The Marine Bill Final Report*, Defra, London.

Adams, V. M., Mills, M., Weeks, R., Segan, D. B., Pressey, R. L., Gurney, G. G., ... & Álvarez-Romero, J. G. (2019). Implementation strategies for systematic conservation planning. *Ambio*, *48*(2), 139-152.

Arkema, K. K., Verutes, G. M., Wood, S. A., Clarke-Samuels, C., Rosado, S., Canto, M., ... & Faries, J. (2015). Embedding ecosystem services in coastal planning leads to better outcomes for people and nature. *Proceedings of the National Academy of Sciences*, *112*(24), 7390-7395.

Armstrong, C. W., N. S. Foley, et al. (2012). "Services from the deep: Steps towards valuation of deep sea goods and services." Ecosystem Services 2: 2-13.

Armstrong, C. W., N. S. Foley, et al. (2014). "Cold water coral reef management from an ecosystem service perspective." Marine Policy 50(Part A): 126-134.

Armstrong, C., Foley, N., Kahui, V., Grehan, A. (2014), Cold water coral reef management from an ecosystem service perspective. Marine Policy (50) 126 - 134.

Armstrong, C., Foley, N., Tinch, R., and Hove, S. (2010). Ecosystem Goods and Services of the Deep Sea, HERMIONE (Hotspot Ecosystem Research and Man's impact on European Seas), Deliverable D6.2:

Armstrong, C.W., Aanesen, M., van Rensburg, T., and Sandorf E.D. (2019b). Willingness to pay for cold water coral protection. Conservation Biology 1523-1739.

Armstrong, C.W., Vondolia, G.K., Foley, N.S., Henry, L., Needham, K. and Ressurreicao, A. (2019a). Expert assessment of risks posed by climate change and anthropogenic activities to ecosystem services in the deep North Atlantic. Frontiers in Marine Science 6: 158

Asch, R. G., & Erisman, B. (2018). Spawning aggregations act as a bottleneck influencing climate change impacts on a critically endangered reef fish. *Diversity and Distributions*, *24*(12), 1712-1728.

Asplund, D., and Eliasson, J. 2016. "Does uncertainty make cost-benefit analyses pointless?" *Transportation Research Part A: Policy and Practice*, *92*, 195-205.

ATLAS Policy Brief "Recognising connectivity and climate change impacts: essential elements for an effective North-Atlantic MPA network"

Austen, M., Anderson, P., Armstrong, C., Döring, R., Hynes, S., Levrel, H., Oinonen, S. and Ressurreição, A. (2019). Valuing Marine Ecosystems - Taking into account the value of ecosystem benefits in the Blue Economy, in Coopman, J., Heymans, J.J., Kellett, P., Muñiz Piniella, A. and French, V. [Eds.] Future Science Brief 5 of the European Marine Board, Ostend, Belgium. 40pp. ISBN: 9789 ISSN: 4920-43696.

Baker, E and Beaudoin, Y (eds) (2013), Deep Sea Minerals and the Green Economy (Vol. 2, Secretariat of the Pacific Community, 2013)

Balmford, A., Gravestock, P., Hockley, N., McClean, C.J., & Roberts, C.M. 2004. The worldwide costs of marine protected areas. PNAS, 101, (26) 9694-9697

Barange, M., Butenschön, M., Yool, A., Beaumont, N., Fernandes, J. A., Martin, A. P., & Allen, J. (2017). The cost of reducing the North Atlantic Ocean biological carbon pump. *Frontiers in Marine Science*, *3*, 290.

Barton, D. 2007. *How much is enough? The value of information from benefit transfers in a policy context. Environmental value transfer: Issues and methods*, Springer: 261-282.

Baxter, W., Corser, T., & Sontag, M. (2019). Application of the natural capital approach to the marine environment to aid decision-making.

Beaumont, N. J., L. Jones, et al. (2014). "The value of carbon sequestration and storage in coastal habitats." Estuarine, Coastal and Shelf Science 137: 32-40.

Beaumont, N. J., Mongruel, R., & Hooper, T. (2017). Practical application of the Ecosystem Service Approach (ESA): lessons learned and recommendations for the future. *International Journal of Biodiversity Science, Ecosystem Services & Management*, *13*(3), 68-78.

Beaumont, N., M. Austen, et al. (2008). "Economic valuation for the conservation of marine biodiversity." Marine Pollution Bulletin 56: 386-396.

Börger, T., Beaumont, N.J., Pendleton, L., Boyle, K.J., Cooper, P., Fletcher, S., Haab, T., Hanemann, M., Hooper, T.L., Hussain, S.S., Portela, R., Stithou, M., Stockill, J., Taylor, T. and Austen, M.C. (2014) Incorporating ecosystem services in marine planning: The role of valuation. Marine Policy, 46. pp. 161-170. ISSN 0308-597X

Börger, T., Böhnke-Henrichs, A., Hattam, C., Piwowarczyk, J., Schasfoort, F., & Austen, M. C. (2018). The role of interdisciplinary collaboration for stated preference methods to value marine environmental goods and ecosystem services. *Estuarine, Coastal and Shelf Science*, *201*, 140-151.

Börger, T., Broszeit, S., Ahtiainen, H., Atkins, J. P., Burdon, D., Luisetti, T., ... & Uyarra, M. C. (2016). Assessing costs and benefits of measures to achieve good environmental status in European regional seas: Challenges, opportunities, and lessons learnt. *Frontiers in Marine Science*, *3*, 192.

Börger, T., Hattam, C., Burdon, D., Atkins, J. P., & Austen, M. C. (2014). Valuing conservation benefits of an offshore marine protected area. *Ecological Economics*, *108*, 229-241.

Bouwma, I., Schleyer, C., Primmer, E., Winkler, K. J., Berry, P., Young, J., ... & Vadineanu, A. (2018). Adoption of the ecosystem services concept in EU policies. *Ecosystem Services*, *29*, 213-222.

Brander, L.M., Bräuer, I., Gerdes, H., Ghermandi, A., Kuik, O., Markandya, A., Navrud, S., Nunes, P.A., Schaafsma, M., Vos, H. and Wagtendonk, A., 2012. Using meta-analysis and GIS for value transfer and scaling up: Valuing climate change induced losses of European wetlands. *Environmental and Resource Economics*, *52*(3), pp.395-413.

Broszeit, S., Beaumont, N. J., Hooper, T. L., Somerfield, P. J., & Austen, M. C. (2019). Developing conceptual models that link multiple ecosystem services to ecological research to aid management and policy, the UK marine example. *Marine pollution bulletin*, *141*, 236-243.

Broszeit, S., Beaumont, N. J., Uyarra, M. C., Heiskanen, A. S., Frost, M., Somerfield, P. J., ... & Austen, M. C. (2017). What can indicators of good environmental status tell us about ecosystem services?: Reducing efforts and increasing cost-effectiveness by reapplying biodiversity indicator data. *Ecological indicators*, *81*, 409-442.

Brouwer, R., Brouwer, S., Eleveld, M. A., Verbraak, M., Wagtendonk, A. J., & van der Woerd, H. J. (2016). Public willingness to pay for alternative management regimes of remote marine protected areas in the North Sea. *Marine Policy*, *68*, 195-204.

Brouwer, R., Langford, I.H., Bateman, I.J., and Turner, R.K. 1999. “A meta-analysis of wetland contingent valuation studies”. *Regional Environmental Change*, *1*(1), 47-57.

Bruno, J. F., Bates, A. E., Cacciapaglia, C., Pike, E. P., Amstrup, S. C., van Hooidonk, R., ... & Aronson, R. B. (2018). Climate change threatens the world’s marine protected areas. *Nature Climate Change*, *8*(6), 499.

Burdon, D., Boyes, S. J., Elliott, M., Smyth, K., Atkins, J. P., Barnes, R. A., & Wurzel, R. K. (2018). Integrating natural and social sciences to manage sustainably vectors of change in the marine environment: Dogger Bank transnational case study. *Estuarine, Coastal and Shelf Science*, *201*, 234-247.

Burkhard, B., Crossman, N., Nedkov, S., Petz, K., & Alkemade, R. (2013). Mapping and modelling ecosystem services for science, policy and practice. Ecosystem Services, 4, 1–3. doi:10.1016/j.ecoser.2013.04.005

Cárcamo, P. F., Garay-Flühmann, R., Squeo, F. A., & Gaymer, C. F. (2014). Using stakeholders' perspective of ecosystem services and biodiversity features to plan a marine protected area. *Environmental Science & Policy*, *40*, 116-131.

Cardno Inc, (2016) An Assessment of the Costs and Benefits of Mining Deep-sea Minerals in the Pacific Island Region. Fiji: Pacific Community, Suva.

Cardoso, A. C., Cochrane, S., Doemer, H., Ferreira, J. G., Galgani, F., Hagebro, C., et al. (2010). *Scientific support to the European Commission on the Marine Strategy Framework Directive. Management Group Report. EUR 24336 EN*. Luxembourg: Joint Research Centre, Office for Official Publications of the European Communities

Carr, M. H., Robinson, S. P., Wahle, C., Davis, G., Kroll, S., Murray, S., ... & Williams, M. (2017). The central importance of ecological spatial connectivity to effective coastal marine protected areas and to meeting the challenges of climate change in the marine environment. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*, 6-29.

Carr, M. H., Saarman, E., & Caldwell, M. R. (2010). The role of "rules of thumb" in science-based environmental policy: California's Marine Life Protection Act as a case study. *Stanford Journal of Law, Science and Policy*, *2*, 1-17.

Cavallo, M., Borja, Á., Elliott, M., Quintino, V. and Touza, J., 2019. Impediments to achieving integrated marine management across borders: The case of the EU Marine Strategy Framework Directive. Marine Policy, 103, pp.68-73

Cavanagh, R. D., Broszeit, S., Pilling, G. M., Grant, S. M., Murphy, E. J., & Austen, M. C. (2016). Valuing biodiversity and ecosystem services: a useful way to manage and conserve marine resources?. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1844), 20161635.

Chiesura, A., and De Groot, R. 2003. "Critical natural capital: a socio-cultural perspective". *Ecological Economics*, *44*(2-3), 219-231.

CICES (2013). "Common International Classification of Ecosystem Services." 2017.

COM (1998) A strategy for Integrating Environment into EU Policies - Cardiff - June 1998 (COM/98/0333 final)

COM (2011) Developing a maritime strategy for the Atlantic Ocean area. COM. (2011).782 Final:

COM (2012) Blue Growth opportunities for marine and maritime sustainable growth. (COM/2012/0494 final)

COM (2013) Action Plan for a Maritime Strategy in the Atlantic Area: delivering smart, sustainable and inclusive growth (COM (2013) 279 final)

COM (2014), Directive 2014/89/EU of the European Parliament and of the Council of 23 July 2014 establishing a framework for maritime spatial planning. http://data.europa.eu/eli/dir/2014/89/oj

Constanza R, d'Arge R, Groots Rd, Farber S, Grasso M, Hannon B, et al. The value of the world's ecosystem services and natural capital. Nature 1997;387:253–60.

Costanza, R et al. "Valuing Ecological Systems and Services." *F1000 Biology Reports* 3 (2011): 14. *PMC*. Web. 5 Feb. 2015.

Costanza, R., R. de Groot, et al. (2017). "Twenty years of ecosystem services: How far have we come and how far do we still need to go?" Ecosystem Services **28**(Part A): 1-16.

Curtis H, Anderson J. Lyme Bay proposed MPA. Indications of social and economic impacts. Sea Fish Industry Authority; 2008.

Curtis, H., & Anderson, J. (2008). Lyme Bay proposed MPA: Indications of social and economic impacts. *Edinburgh: Sea Fish Industry Authority, Submission for Marine and Fisheries Agency*.

D'Aloia, C. C., Daigle, R. M., Côté, I. M., Curtis, J. M., Guichard, F., & Fortin, M. J. (2017). A multiple-species framework for integrating movement processes across life stages into the design of marine protected areas. *Biological Conservation*, *216*, 93-100.

Danovaro, R., A., Dell’Anno, Corinaldesi, C., Magagnini, M., Noble, R., Tamburini, C., Weinbauer, M., 2008a. Major viral impact on the functioning of benthic deepsea ecosystems. Nature 454 (7208), 1084–1087.

Danovaro, R., Gambi, C., Dell’Anno, A., Corinaldesi, C., Fraschetti, S., Vanreusel, A., Vincx, M., Gooday, A.J., 2008b. Exponential decline in deep sea ecosystem functioning linked to benthic biodiversity loss. Current Biology 18, 1–8

d'Aspremont, C., and Gevers, L. 2002. “Social welfare functionals and interpersonal comparability”. *Handbook of Social Choice and Welfare*, *1*, 459-541.

de Groot, R., L. Brander, et al. (2012). "Global estimates of the value of ecosystems and their services in monetary units." Ecosystem Services 1(1): 50-61.

Defra 2009, *Marine and coastal access bill impact assessment: Introduction to the House of Commons*, Defra, London.

Defra. Impact assessment of measures to protect marine biodiversity in Lyme Bay. Department for Environment, Farming and Rural Affairs; 2008.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., ... & Polasky, S. (2018a). Assessing nature's contributions to people. *Science*, *359*(6373), 270-272.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R. T., Molnár, Z., ... & Polasky, S. (2018b). RE: There is more to Nature’s Contributions to People than Ecosystem Services–A response to de Groot et al. *Science*, *359*, 3.

Douvere, F., & Ehler, C. N. (2011). The importance of monitoring and evaluation in adaptive maritime spatial planning. *Journal of Coastal Conservation*, *15*(2), 305-311.

Edgar, G. J., Stuart-Smith, R. D., Willis, T. J., Kininmonth, S., Baker, S. C., Banks, S., ... & Buxton, C. D. (2014). Global conservation outcomes depend on marine protected areas with five key features. *Nature*, *506*(7487), 216.

EEA (2016) progress report on mapping and assessing the condition of Europe's ecosystems eftec (2010) "Benefits and costs of conserving biodiversity and ecosystem services". Final Report to the European Commission, DG Environment. ENV/07.0307/2009/553444/ETU/B2.

Ehler, C., and F. Douvere. 2009. *Marine Spatial Planning: A Step-by-Step Approach toward Ecosystem-based Management.* Intergovernmental Oceanographic Commission and Man and Biosphere Programme. IOC Manual and Guides No 53, ICAM Dossier No 6. Paris: UNESCO.

European Commission Directive 2008/56/EC, changed through Commission Directive (EU) 2017/845 and Commission Decision (EU) 2017/848

European Commission Directive 2014/89/EU establishing a framework for maritime spatial planning. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32014L0089

European Marine Board (2019), Future Science Brief n°5: Valuing Marine Ecosystem Services

Evans, M. C., Davila, F., Toomey, A., & Wyborn, C. (2017). Embrace complexity to improve conservation decision making. *Nature ecology & evolution*, *1*(11), 1588.

Farber, S. C., Costanza, R., & Wilson, M. A. (2002). Economic and ecological concepts for valuing ecosystem services. *Ecological economics*, *41*(3), 375-392.

Farley, J. (2008). The role of prices in conserving critical natural capital. Conservation Biology, 22(6), 1399-1408.

Fischer, P., Kastenmüller, A., Greitemeyer, T., Fischer, J., Frey, D., & Crelley, D. (2011). Threat and selective exposure: The moderating role of threat and decision context on confirmatory information search after decisions. *Journal of Experimental Psychology: General*, *140*(1), 51.

Flyvbjerg, B. 2009. "Survival of the unfittest: why the worst infrastructure gets built — and what we can do about it". *Oxford Review of Economic Policy*, *25*(3), 344–367.

Foley, N. S., R. Corless, et al. (2014). "Developing a Comparative Marine Socio-Economic Framework for the European Atlantic Area." Journal of Ocean and Coastal Economics **2014**(1 (3)).

Foley, N., T. M. Van Rensburg, et al. (2010). "The ecological and economic value of cold-water coral ecosystems." Ocean & Coastal Management **53**(7): 313-326.

Galparsoro, I., A. Borja, et al. (2014). "Mapping ecosystem services provided by benthic habitats in the European North Atlantic Ocean." Frontiers in Marine Science **1**(23).

Gell, F. R., & Roberts, C. M. (2003). Benefits beyond boundaries: the fishery effects of marine reserves. *Trends in Ecology & Evolution*, *18*(9), 448-455.

Glenn, H., Wattage, P., Mardle, S., Rensburg, T. Van, Grehan, A. and Foley, N. (2010). Marine protected areas-substantiating their worth. Marine Policy 34: 421–430.

Global Ocean Commission. (2014). From decline to recovery: A rescue package for the global ocean. *Global Ocean Commission*.

Grimsrud, K., Lindhjem, H., Barton, D. N., & Navrud, S. (2018). Challenges to ecosystem service valuation for wealth accounting. *Inclusive Wealth Report 2018*, 136.

Haines-Young, R., & Potschin, M. (2013). Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012. EEA Framework Contract No EEA. *Contract No EEA/IEA/09/003*.

Halpern, B.S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecol. Appl. 13: 117–137

Hanley, N., Hynes, S., Patterson, D., & Jobstvogt, N. (2015). Economic valuation of marine and coastal ecosystems: is it currently fit for purpose?. *Journal of Ocean and Coastal Economics*.

Hart, P. S., & Nisbet, E. C. (2012). Boomerang effects in science communication: How motivated reasoning and identity cues amplify opinion polarization about climate mitigation policies. *Communication Research*, *39*(6), 701-723.

Hartin, C. A., Bond-Lamberty, B., Patel, P., & Mundra, A. (2016). Ocean acidification over the next three centuries using a simple global climate carbon-cycle model: projections and sensitivities. *Biogeosciences*, *13*(15), 4329-4342.

Hattam, C., Böhnke-Henrichs, A., Börger, T., Burdon, D., Hadjimichael, M., Delaney, A., ... & Austen, M. C. (2015). Integrating methods for ecosystem service assessment and valuation: Mixed methods or mixed messages?. *Ecological Economics*, *120*, 126-138.

Hoagland, P., & Jin, D. (2008). Accounting for marine economic activities in large marine ecosystems. *Ocean & Coastal Management*, *51*(3), 246-258.

Hoch, S.J., and Loewenstein, G.F. 1991. "Time-inconsistent preferences and consumer self-control". *Journal of Consumer Research*, *17*(4), 492-507

Hoegh-Guldberg, O., & Bruno, J. F. (2010). The impact of climate change on the world's marine ecosystems. *Science*, *328*(5985), 1523-1528.

Homarus Ltd (2007) Estimate of Economic Values of Activities in Proposed Conservation Zone in Lyme Bay – a very localised example with opportunity costs and benefits of a protected area.

Hönisch, B., Hemming, N. G., Archer, D., Siddall, M., & McManus, J. F. (2009). Atmospheric carbon dioxide concentration across the mid-Pleistocene transition. *Science*, *324*(5934), 1551-1554.

Hooper, T., Börger, T., Langmead, O., Marcone, O., Rees, S. E., Rendon, O., ... & Austen, M. (2019). Applying the natural capital approach to decision making for the marine environment. *Ecosystem Services*, *38*, 100947.

Hooper, T., Langmead, O., & Ashley, M. (2016). Ecosystem Services in Marine Environmental Impact Assessment: tools to support marine planning at project and strategic scales. In *Handbook on Biodiversity and Ecosystem Services in Impact Assessment*. Edward Elgar Publishing.

Howard, J., McLeod, E., Thomas, S., Eastwood, E., Fox, M., Wenzel, L., & Pidgeon, E. (2017). The potential to integrate blue carbon into MPA design and management. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *27*, 100-115.

Hussain, S.S., Winrow-Giffin, A., Moran, D., Robinson, L.A., Fofana, A., Paramor, O.A.L., & Frid, C.L.J. 2010. An ex-ante ecological economic assessment of the benefits arising from marine protected areas designation in the UK. *Ecological Economics*, 69, (4) 828-838

IPBES (2017). Update on the classification of nature's contributions to people by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. . Bonn, Germany.

Jobstvogt N, Townsend M, Witte U, Hanley N (2014) How Can We Identify and Communicate the Ecological Value of Deep-Sea Ecosystem Services? PLoS ONE 9(7): e100646. doi:10.1371/journal.pone.0100646

Jobstvogt, N., Hanley, N., Hynes, S., Kenter, J., & Witte, U. (2014). Twenty thousand sterling under the sea: estimating the value of protecting deep-sea biodiversity. *Ecological Economics*, *97*, 10-19.

Johnson, D., Ferreira, M. A., & Kenchington, E. (2018). Climate change is likely to severely limit the effectiveness of deep-sea ABMTs in the North Atlantic. *Marine Policy*, *87*, 111-122.

Johnson, D., M. A. Ferreira, et al. (2017). Report on priorities for an expert assessment of North Atlantic MPAs, EBSAs, and VMEs in ABNJ. Contribution to the EU H2020 ATLAS project.

Kingdon, J. W., & Thurber, J. A. (1984). *Agendas, alternatives, and public policies* (Vol. 45, pp. 165-169). Boston: Little, Brown.

Knight, A. T., R. M. Cowling, A. F. Boshoff, S. L. Wilson, and S. M. Pierce. 2011. Walking in STEP: Lessons for linking spatial prioritisations to implementation strategies. Biological Conservation **144**:202-211.

Knight, A. T., R. M. Cowling, and B. M. Campbell. 2006. An operational model for implementing conservation action. Conservation Biology **20**:408-419.

Knight, A. T., R. M. Cowling, M. Rouget, A. Balmford, A. T. Lombard, and B. M. Campbell. 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. Conservation Biology **22**:610-617.

Laffoley D, Maltby E, Vincent MA, Mee L, Dunn E, Gilliland P, et al. The ecosystem approach. Coherent actions for marine and coastal environments. A report to the UK government. English Nature; 2004.

Landers, D.H, and Nahlik, A.M. (2013). Final Ecosystem Goods and Services Classification System (FEGS-CS). EPA/600/R-13/ORD-004914. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Laville, F. 2000. “Foundations of procedural rationality: cognitive limits and decision processes.” *Economics & Philosophy*, *16*(1), 117-138.

Lawton, J. (2010). *Making Space for Nature: a review of England's wildlife sites and ecological networks*. Defra.

Liquete, C., Piroddi, C., Drakou, E. G., Gurney, L., Katsanevakis, S., Charef, A., & Egoh, B. (2013). Current status and future prospects for the assessment of marine and coastal ecosystem services: a systematic review. *PloS one*, *8*(7), e67737.

Lopes, R., & Videira, N. (2013). Valuing marine and coastal ecosystem services: an integrated participatory framework. *Ocean & Coastal Management*, *84*, 153-162.

Luisetti, T., Turner, K., & Bateman, I. 2008, An ecosystem services approach to assess managed realignment coastal policy in England. CSERGE Working Paper ECM 08-04

Luisetti, T., Turner, R. K., Jickells, T., Andrews, J., Elliott, M., Schaafsma, M., ... & Watts, W. (2014). Coastal zone ecosystem services: from science to values and decision making; a case study. *Science of the Total Environment*, *493*, 682-693.

MA (2005). Ecosystems and Human Well-being: Sythesis. Millennium Ecosystem Assessment. Washington DC, Island Press.

Mace, G. M. (2014). Whose conservation?. *Science*, *345*(6204), 1558-1560.

Mackie, P., and Preston, J. 1998. “Twenty-one sources of error and bias in transport project appraisal”. *Transport Policy*, *5*(1), 1-7.

Magris, R. A., Pressey, R. L., Weeks, R., & Ban, N. C. (2014). Integrating connectivity and climate change into marine conservation planning. *Biological Conservation*, *170*, 207-221.

March, J.G., and Simon, H.A. 1958. “Cognitive limits on rationality”. *Organizations*, 136-171.

Marre, J. B. (2014). *L'évaluation économique des services écosystémiques marins et côtiers et son utilisation dans la prise de décision: cas d'étude en Nouvelle-Calédonie et en Australie* (Doctoral dissertation, Brest).

Marre, J. B., Thébaud, O., Pascoe, S., Jennings, S., Boncoeur, J., & Coglan, L. (2016). Is economic valuation of ecosystem services useful to decision-makers? Lessons learned from Australian coastal and marine management. *Journal of Environmental Management*, *178*, 52-62.

McKinley, E., Acott, T., & Stojanovic, T. (2019). Socio-cultural dimensions of marine spatial planning. In *Maritime spatial planning* (pp. 151-174). Palgrave Macmillan, Cham.

McLeod, E., Salm, R., Green, A., & Almany, J. (2009). Designing marine protected area networks to address the impacts of climate change. *Frontiers in Ecology and the Environment*, *7*(7), 362-370.

McVittie, A. & Moran, D. 2008, *Determining monetary values for use and non-use goods and services: Marine Biodiversity – primary valuation, Final Report*, Defra, London.

Milon, J. W., & Alvarez, S. (2019). The Elusive Quest for Valuation of Coastal and Marine Ecosystem Services. *Water*, *11*(7), 1518.

Moon, K., Adams, V. M., JANUCHOWSKI-HARTLEY, S. R., Polyakov, M., Mills, M., Biggs, D., ... & Raymond, C. M. (2014). A multidisciplinary conceptualization of conservation opportunity. *Conservation Biology*, *28*(6), 1484-1496.

Moran, D., Hussain, S. S., Fofana, A., Frid, C. L. J., Paramor, O. A. L., Robinson, L. A., & Winrow-Giffin, A. 2008, *The Marine Bill – Marine Nature Conservation Proposals –Valuing the Benefits*, London, Defra.

Mulazzani, L. and G. Malorgio (2017). "Blue growth and ecosystem services." Marine Policy **85**(Supplement C): 17-24.

Navrud, S., Lindhjem, H., & Magnussen, K. (2017). Valuing marine ecosystem services loss from oil spills for use in cost–benefit analysis of preventive measures. In *Handbook on the economics and management of sustainable oceans*. Edward Elgar Publishing.

Nordic Council of Ministers (2017) Ecosystem services in MSP: Ecosystem services approach as a common Nordic understanding for MSP

Norton, D. and Hynes, S. (2018). Estimating the Benefits of the Marine Strategy Framework Directive in Atlantic Member States: A Spatial Value Transfer Approach, Ecological Economics, 151, 82–94

Norton, D., Hynes, S. and J. Boyd (2018). Valuing Ireland's Coastal, Marine and Estuarine Ecosystem Services. EPA Report (2014-NC-MS-1). Wexford, Ireland, EPA.

Oinonen, S., Börger, T., Hynes, S., Buchs, A. K., Heiskanen, A. S., Hyytiäinen, K., ... & van der Veeren, R. (2016). The role of economics in ecosystem based management: the case of the EU Marine Strategy Framework Directive; first lessons learnt and way forward. *Journal of Ocean and Coastal Economics*.

Pearce DW, Turner RK. Economics of natural resources and the environment. Hemel Hempstead: Harvester Wheatsheaf; 1990

Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I. C., ... & Falconi, L. (2017). Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science*, *355*(6332), eaai9214.

Pecl, G. T., Araújo, M. B., Bell, J. D., Blanchard, J., Bonebrake, T. C., Chen, I. C., ... & Falconi, L. (2017). Biodiversity redistribution under climate change: Impacts on ecosystems and human well-being. *Science*, *355*(6332), eaai9214.

Pelletier, É. (2018). Les aires marines protégées : evolution récente et perspectives de développement. *Le Naturaliste canadien*, *142*(2), 167–181. doi:10.7202/1047157ar

Pendleton, L., Mongruel, R., Beaumont, N., Hooper, T., & Charles, M. (2015). A triage approach to improve the relevance of marine ecosystem services assessments. *Marine Ecology Progress Series*, *530*, 183-193.

Pfeffer, J., and R. I. Sutton. 1999. Knowing "What" to Do Is Not Enough: Turning Knowledge into Action. California Management Review **42**:83-108.

Pielke Jr, R. A. (2007). *The honest broker: making sense of science in policy and politics*. Cambridge University Press.

Ramirez-Llodra, E., Tyler, P. A., Baker, M. C., Bergstad, O. A., Clark, M. R., Escobar, E., ... & Van Dover, C. L. (2011). Man and the last great wilderness: human impact on the deep sea. *PLoS one*, *6*(8), e22588.

Ravenscroft, N. 2019. A new normative economics for the formation of shared values. *Sustainability Science* DOI: 10.1007/s11625-018-0652-4

Rees, S. E., Attrill, M. J., Austen, M. C., Mangi, S. C., Richards, J. P., & Rodwell, L. D. (2010). Is there a win–win scenario for marine nature conservation? A case study of Lyme Bay, England. *Ocean & Coastal Management*, *53*(3), 135-145.

Ressurreição, A., Gibbons, J., Kaiser, M., Dentinho, T. P., Zarzycki, T., Bentley, C., ... & Edwards-Jones, G. (2012). Different cultures, different values: The role of cultural variation in public's WTP for marine species conservation. *Biological Conservation*, *145*(1), 148-159.

Ressurreição, A., J. Gibbons, et al. (2011). "Economic valuation of species loss in the open sea." Ecological Economics 70(4): 729-739.

Ressurreição, A., Zarzycki, T., Kaiser, M., Edwards-Jones, G., Dentinho, T. P., Santos, R. S., & Gibbons, J. (2012). Towards an ecosystem approach for understanding public values concerning marine biodiversity loss. *Marine Ecology Progress Series*, *467*, 15-28.

Robbins L. (1935). An Essay on the Nature and Significance of Economic Science. London: MacMillan and Co

Roberts, C. M., O'Leary, B. C., McCauley, D. J., Cury, P. M., Duarte, C. M., Lubchenco, J., ... & Worm, B. (2017). Marine reserves can mitigate and promote adaptation to climate change. *Proceedings of the National Academy of Sciences*, *114*(24), 6167-6175.

Rose, D. C., Mukherjee, N., Simmons, B. I., Tew, E. R., Robertson, R. J., Vadrot, A. B., ... & Sutherland, W. J. (2017). Policy windows for the environment: tips for improving the uptake of scientific knowledge. *Environmental Science & Policy*.

Rose, D. C., Sutherland, W. J., Amano, T., González-Varo, J. P., Robertson, R. J., Simmons, B. I., ... & Wu, W. (2018). The major barriers to evidence-informed conservation policy and possible solutions. *Conservation letters*, *11*(5), e12564.

Russ, G.R. et al. 2004. Marine reserve benefits local fisheries. Ecol. Appl. 14: 597–606.

Saarman, Emily, Mary Gleason, John Ugoretz, Satie Airamé, Mark Carr, Evan Fox, Adam Frimodig, Tom Mason, and Jason Vasques. "The role of science in supporting marine protected area network planning and design in California." *Ocean & Coastal Management* 74 (2013): 45-56.

Sagebiel, J., Schwartz, C., Rhozyel, M., Rajmis, S., & Hirschfeld, J. (2016). Economic valuation of Baltic marine ecosystem services: blind spots and limited consistency. *ICES Journal of Marine Science*, *73*(4), 991-1003.

Sala, E., Lubchenco, J., Grorud-Colvert, K., Novelli, C., Roberts, C., & Sumaila, U. R. (2018). Assessing real progress towards effective ocean protection. *Marine Policy*, *91*, 11-13.

Salomidi, M., Katsanevakis, S., Borja, A., Braeckman, U., Damalas, D., Galparsoro, I., Mifsud, R., Mirto, S., Pascual, M., Pipitone, C., Rabaut, M., Todorova, V., Vassilopoulou, V. and Vega Fernandez, T. (2012). Assessment of goods and services, vulnerability, and conservation status of European seabed biotopes: a stepping stone towards ecosystem-based marine spatial management. Mediterranean Marine Science, 13(1): 49-88.

Sarkki, Simo, Rob Tinch, Jari Niemelä, Ulrich Heink, Kerry Waylen, Johannes Timaeus, Juliette Young, Allan Watt, Carsten Neßhöver, and Sybille van den Hove. "Adding ‘iterativity’to credibility, relevance, legitimacy: A novel scheme to highlight dynamic aspects of science–policy interfaces." Environmental Science & Policy (2015).

Slovic, P. 1995. “The construction of preference”. *American Psychologist*, *50*(5), 364.

Smith, J., & Metaxas, A. (2018). A decision tree that can address connectivity in the design of Marine Protected Area Networks (MPAn). *Marine Policy*, *88*, 269-278.

Stithou, M. (2017). Considerations of Socio-Economic Input, Related Challenges and Recommendations for Ecosystem-Based Maritime Spatial Planning: A Review. *Journal of Ocean and Coastal Economics*, *4*(1), 1.

Sumaila, U.R., D. Zeller, R. Watson, J. Alder & D. Pauly. 2007. Potential costs and benefits of marine reserves in the high seas. Mar. Ecol. Prog. Ser. 345: 305–310.

Sutherland, W. J., & Wordley, C. F. (2017). Evidence complacency hampers conservation. *Nature ecology & evolution*, *1*(9), 1215.

TEEB (2010). The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB: 29.

TEEB (2012) “Why Value the Oceans?” p23. http://www.teebweb.org/media/2013/10/2013-Why-Value-the-Oceans-Discussion-Paper.pdf

Thurber, A. R., A. K. Sweetman, et al. (2014). "Ecosystem function and services provided by the deep sea." Biogeosciences 11(14): 3941-3963.

Tinch, R, Cindy Schoumacher, and Sybille van den Hove. (2015) "Exploring barriers to integration of biodiversity concerns across EU policy." in Gasparatos, A., & Willis, K. J. (eds.). Biodiversity in the Green Economy. Routledge.

Tinch, R., C. Armstrong, et al. (2011). Report on policy demands for value evidence, Deliverable 6.3 of HERMIONE project: 45.

van den Hove, S. and V. Moreau (2007). Deep-Sea Biodiversity and Ecosystems: A scoping report on their socio-economy, management and governance, UNEP.

Van Kerkhoff, L., & Lebel, L. (2006). Linking knowledge and action for sustainable development. *Annu. Rev. Environ. Resour.*, *31*, 445-477.

Veidemane, K., Ruskule, A., Strake, S., Purina, I., Aigars, J., Sprukta, S., ... & Klepers, A. (2017). Application of the marine ecosystem services approach in the development of the maritime spatial plan of Latvia. *International Journal of Biodiversity Science, Ecosystem Services & Management*, *13*(1), 398-411.

Wainger, Lisa A. Wainger, Dennis M. King, Richard N. Mack, Elizabeth W. Price, Thomas Maslin, 2010, Can the concept of ecosystem services be practically applied to improve natural resource management decisions?, Ecological Economics, 69 : 978-987

Wakefield, J. R., & Myers, K. (2016). Social cost benefit analysis for deep sea minerals mining. Marine Policy.

Wattage, P., Glenn, H., Mardle, S., Van Rensburg, T., Grehan, A. and Foley, N., (2011). Economic value of conserving deep-sea corals in Irish waters: A choice experiment study on marine protected areas.

Woodson, C. B., Micheli, F., Boch, C., Al-Najjar, M., Espinoza, A., Hernandez, A., ... & Torre, J. (2019). Harnessing marine microclimates for climate change adaptation and marine conservation. *Conservation Letters*, *12*(2), e12609.

Wynne, B. (2007). Dazzled by the mirage of influence? STS-SSK in multivalent registers of relevance. *Science, Technology, & Human Values*, *32*(4), 491-503.

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