

The benefits to people of expanding Marine Protected Areas

Final report

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Executive Summary

- This study focuses on how the economic value of marine ecosystem services to people and communities is expected to change with the expansion of marine protected areas (MPAs). It is recognised, however, that instrumental economic value derived from ecosystem services is only one component of the overall value of the marine environment and that the intrinsic value of nature also provides an argument for the conservation of the marine habitats and biodiversity.
- The main objective of this study is to evaluate the economic case for MPAs through an assessment of the costs and benefits of expanding 'no-take' MPAs.
- An MPA is a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (IUCN-WCPA, 2008). When they are well designed and managed, MPAs allow for the protection and restoration of key habitats, the replenishment of fish stocks and can enhance the resilience of marine ecosystems.
- The study develops a set of six mapped scenarios for the global expansion of MPAs. The scenarios vary along two dimensions: 1. the coverage of MPAs as a proportion of total marine area; 2. the characteristics of target locations for MPAs in terms of biodiversity and degree of human impact. The scenarios are explorative: they pose the question "what if MPAs were expanded in this way?"
- The results of the cost-benefit analysis show that all six scenarios for expanding MPAs to 10% and 30% coverage are economically advisable. The ratios of benefits to costs are in the range 3.17 – 19.77. In the case of the scenario that achieves 10% coverage of total marine area and targets areas with high biodiversity and low human impact, each dollar invested yields a return of around 20 dollars in benefits.
- Net benefits continue to accumulate as the area of protection increases up to 30% which is the extent of this analysis. The rate at which net benefits accrue, however, slows as the area of MPA coverage increases.
- The total cost of achieving 10% coverage of MPAs is estimated in the range of USD 45-47 billion over the period 2015-2050.¹ The total costs of achieving 30% coverage are in the range USD 223-228 billion. The cost categories included in these estimates are the set-up and operating costs of MPAs and the opportunity costs to commercial fisheries. The costs vary depending on the size and location of MPAs. Set-up and operating costs, expressed per unit of area protected, decrease with the scale of MPA coverage, whereas opportunity costs to fisheries increase.
- The total ecosystem service benefits of achieving 10% coverage of MPAs is estimated in the range USD 622-923 billion over the period 2015-2050; and for 30% coverage, the benefits range between USD 719-1,145 billion. The ecosystem services covered in the estimated benefits include coastal protection, fisheries, tourism, recreation and carbon storage provided by coral reefs, mangroves and coastal wetlands. Variation in benefits across scenarios is largely due to differences in the provision of services from coral reefs.

¹ All monetary values are expressed as present values computed over the period 2015-2050 using a discount rate of 3% in USD at 2013 price levels.

- The analysis contains only a partial view of the full set of costs and benefits associated with expanding MPAs. On the costs side we are missing information on the opportunity costs of other marine activities such as mineral extraction and energy generation. On the benefit side we are missing information on the potentially positive impacts of MPAs on some ecosystems (e.g. seamounts, seagrass, kelp forests) and ecosystem services (e.g. bio-prospecting and existence values associated with marine biodiversity). On balance, we expect that adding further information would tend to increase the benefits of expansion relative to costs since existing estimates for non-use values for marine biodiversity are generally high.
- Substantial knowledge gaps exist regarding how MPAs affect ecosystems and the provision of ecosystem services. Further research is required to fill these gaps and allow a more comprehensive assessment of MPA costs and benefits.
- A MPA network approach is likely to yield more benefits for species, habitats and humans than the sum of its parts. Due to considerable data limitations, however, this study does not examine network effects of MPAs and the management measures that ensure effectiveness. In designing and designating MPAs, the social and ecological perspectives (including connectivity of species and habitats in a network approach) also need to be taken into consideration.
- The adoption of MPAs should not become an excuse for not implementing other recommended management measures. MPAs are an essential element of the “management tools mosaic” but should not be treated as the panacea. In practice MPAs will be used to reinforce and strengthen other forms of management and complement other types of intervention.
- A set of case studies is used to highlight issues that cannot be addressed in the global assessment of MPA expansion, including the distribution of the costs and benefits of MPAs across stakeholders, the importance of marine resources to coastal communities, the social impacts of MPAs, non-use values of biodiversity conservation, and indirect employment effects in other sectors.

1 Introduction

1.1 Marine ecosystem services

The oceans and coastal ecosystems are vital to life on Earth in terms of the provision of ecosystem services. Ecosystem services are the benefits that ecosystems provide for people (MA, 2005) and include both the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010). Marine ecosystem services include seafood, genetic material, coastal protection, carbon sequestration, biodiversity, recreation and other cultural services (Beaumont *et al.* 2007; Bohnke-Henrichs *et al.*, 2013).²

Marine ecosystem services have high economic values in terms of their contribution to specific sectors of the economy (e.g. fisheries, tourism) and to human welfare (de Groot *et al.*, 2012). Specific marine ecosystems provide multiple services; for example, coral reefs may provide coastal protection (van Zanten *et al.*, 2014; Ferrario *et al.*, 2014), support fisheries and provide recreational opportunities (Brander *et al.*, 2007). Similarly, mangroves and other coastal ecosystems store substantial quantities of organic carbon (Murray *et al.*, 2011; Pendleton *et al.*, 2012), mitigate storm damage (Barbier *et al.* 2011) and function as a nursery for fisheries. The high seas are also recognised to provide a wide range of ecosystem services (Rogers *et al.*, 2013; Armstrong *et al.*, 2014) and seamounts have been identified as hotspots of pelagic biodiversity (Morato *et al.*, 2010).

This study focuses on how the economic value of marine ecosystem services to people and communities is expected to change with the expansion of marine protected areas. It is recognised, however, that instrumental economic value derived from ecosystem services is only one component of the overall value of the marine environment (Turner, 1999) and that the intrinsic value of nature also provides an argument for the conservation of the marine habitats and biodiversity (Balmford *et al.*, 2011).

1.2 Threats to marine ecosystems

Marine ecosystems face a wide range of threats including land and marine based pollution, eutrophication, infrastructure development (leading to habitat loss and degradation), sedimentation, over fishing, hypoxia (de-oxygenation), invasive species, acidification and changes in temperature, currents and sea level (Brander, 2007; Turley *et al.*, 2013; Noone *et al.* 2014).

Some threats to the marine environment, such as overfishing and habitat loss, are widely researched and increasingly well understood. Other less visible threats, such as ocean acidification and hypoxia, are only now emerging as important issues that need to be addressed. Moreover, the interaction and cumulative effects of multiple stressors are highly complex and the combined impact on ecosystems and the provision of services is largely unknown or highly uncertain (Noone *et al.*, 2014; Brander, 2015).

Ocean and coastal ecosystems have generally lacked effective management or protection from overuse and other threats. This is particularly the case where they cross borders or lie outside of national jurisdictions (Rogers *et al.*, 2013).

² Bohnke-Henrichs *et al.* (2013) develop a comprehensive classification of marine ecosystem services that identifies 21 services grouped into four categories: provisioning, regulating, habitat services, and cultural and amenity services.

Overall assessments of the health of the marine environment and associated provision of ecosystem services generally show negative trends (Burke *et al.*, 2011; Halpern *et al.*, 2012; Brander *et al.*, 2012a; Noone *et al.*, 2014; Hoegh-Guldberg *et al.*, 2015).

1.3 Marine protected areas as a potential solution

In response to increasing degradation of the marine environment and declining provision of ecosystem services, several international policy fora as well as locally run initiatives have called for and initiated the development of marine protected areas (MPAs). An MPA is a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (IUCN-WCPA, 2008). MPAs may be organised in networks to enhance conservation and other objectives through cooperation and synergies. A MPA network approach is likely to yield more benefits for species, habitats and humans than the sum of its parts (Hugenholtz, 2008).³

MPAs include a diverse variety of management systems and restrictions on economic activities, which have accordingly been assigned diverse titles (IUCN, 2008). MPAs can be grouped broadly into two categories: areas of full protection in which all removals of resources are strictly prohibited and areas of partial protection that allow various moderated economic activities. The former group includes 'no-take MPAs', 'marine reserves' and 'marine conservation zones', in which activities including fishing, aquaculture, water transportation and industrial development are prohibited (Jones, 2008; Marinesque *et al.*, 2012; NRC, 2001). The latter group includes 'multiple-use MPAs', 'marine parks' and 'habitat/species management areas', which are designed to achieve diverse objectives, including biodiversity conservation, protection of cultural heritage, enhancement of sustainable use of resources, and comprise sites of varying degrees of protection (ANZECC TFMPA, 1998; Horigue *et al.*, 2012; USNMPAC, 2006; Agardy, 2000; Davis *et al.*, 2004; Kelleher, 1999; Ovetz, 2006; Teh *et al.*, 2012). For clarity, we use the term 'no-take MPA' for areas of full protection, and 'multiple-use MPA' for areas of partial protection. Future expansion of MPAs is expected to include both no-take and multiple use MPAs. The analysis described in this report, however, focuses on the expansion of no-take MPAs only since we are unable to assess the impacts of multiple-use MPAs at a global scale.

The coverage of multiple-use MPAs has far exceeded that of fully protected no-take MPAs (Wood *et al.*, 2008). Currently, about 3.4% of marine area is designated as MPA, with 0.59% established as no-take MPAs (Thomas *et al.*, 2014). Detailed information on global patterns and trends in MPA development can be found in Fox *et al.* (2012). The location and extent of existing MPAs under varying levels of protection is shown in Figure 1.

³ We are unable to examine MPA network effects within the scope of this study. Examples of MPA networks are given in the case studies on Fiji locally managed marine areas (section 5.2) and the Coral Triangle Initiative (section 5.8).

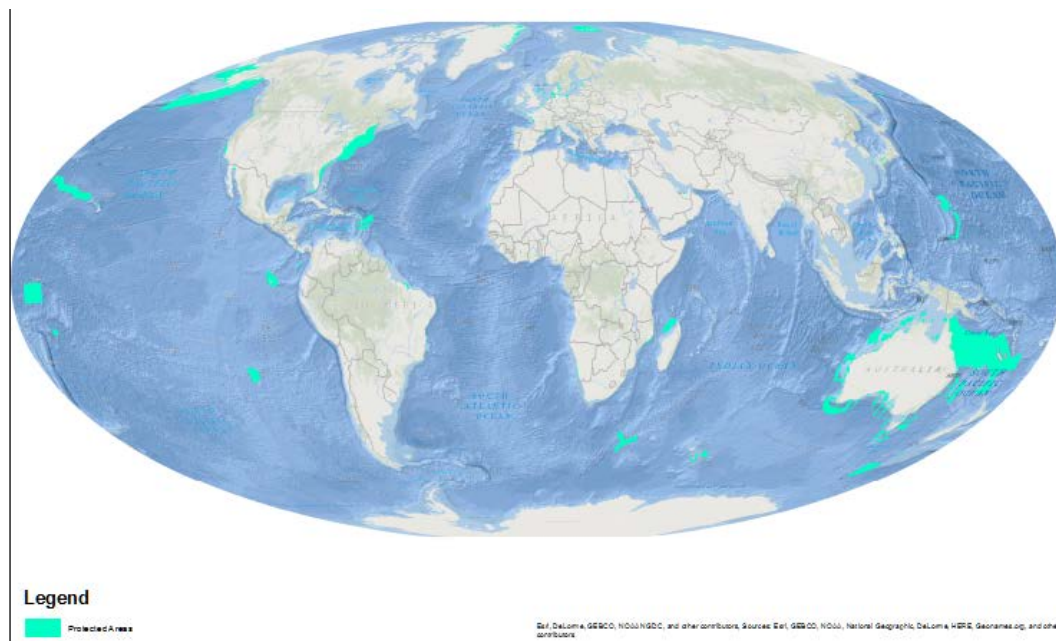


Figure 1 Location of marine protected areas (October 2014)

MPAs are increasingly used for managing human activities in the marine environment (Bohnsack, 1993, 1998; Ludwig *et al.*, 1993; Mangel, 2000; Yagi *et al.*, 2010). When they are well designed and managed, MPAs allow for the protection and restoration of key habitats, the replenishment of fish stocks and can enhance the resilience of marine ecosystems (Salm *et al.*, 2000). In doing so, they may increase the provision of some ecosystem services such as recreation and tourism, coastal protection and carbon sequestration. In addition, by increasing fish biomass, size, density and species richness within MPAs (Lester *et al.* 2009; Halpern *et al.* 2012), they may sustain or increase yields of nearby fisheries through exporting fish larvae and adults (spill-over and recruitment effects). MPAs can be used to ameliorate the negative impacts of human activities such as overfishing, oil and mineral extraction, aggregate mining and discharge of waste-water (Gaines *et al.*, 2010; Hastings and Botsford, 1999; O'Leary *et al.*, 2012; Wells *et al.*, 2007).

The two predominant statements calling for the global expansion of networks of MPAs are the Durban Action Plan (WPC, 2003) and the Convention on Biological Diversity (CBD) Aichi Target 11.⁴

The Durban Action Plan, developed at the 2003 Vth IUCN World Parks Congress, calls to:

"Establish by 2012 a global system of effectively managed, representative networks of marine and coastal protected areas, consistent with international law and based on scientific information, that: a. Greatly increases the marine and coastal area managed

⁴ More recently, the 'Promise of Sydney' statement issued by the 2014 World Parks Congress recommends: "Urgently increase the ocean area that is effectively and equitably managed in ecologically representative and well-connected systems of MPAs or other effective conservation measures. This network should target protection of both biodiversity and ecosystem services and should include at least 30% of each marine habitat. The ultimate aim is to create a fully sustainable ocean, at least 30% of which has no extractive activities."

in marine protected areas by 2012; these networks should be extensive and include strictly protected areas that amount to at least 20-30% of each habitat, and contribute to a global target for healthy and productive oceans;"

The Convention on Biological Diversity (CBD) Aichi Target 11, adopted in 2010 at the 10th Conference of the Parties in Nagoya, Japan, requires that:

"By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes."

Progress has been made towards meeting these targets but considerably more needs to be done in order to ensure the effectiveness and ecological representativeness of MPAs, in addition to the geographic coverage (Ban *et al.*, 2014; Bignoli *et al.*, 2014; Dunn *et al.*, 2014; Fox *et al.*, 2014).

2 Objectives

The main objective of this study is to evaluate whether there is an economic case for expanding MPAs through an assessment of the costs and benefits of protecting marine habitats using no-take MPAs. This study aims to assess the net benefits of additional protection and is not an analysis of the total benefits of marine ecosystem services.

The specific objectives are to:

- Develop six **global scenarios** for the location of new and expanded no-take MPAs that are effectively managed. The scenarios are described along two dimensions. First, based on the proportion of marine area designated as MPA (e.g. 10% under the CBD target⁵ and 30% under the Durban target). Second, based on the spatial location of MPAs, targeting areas of high biodiversity or areas facing the highest anthropogenic pressures.
- Assess the **additional benefits** of creating and expanding effectively managed MPAs in terms of the economic value of changes in the provision of ecosystem services.
- Assess the **additional costs** of establishing and operating MPAs, including the costs of effective management
- Estimate the **net benefits** in economic terms (i.e. benefits minus costs)
- Assess the impact of expanding MPAs on **employment**
- Present a set of **case studies** of existing MPAs or regional networks to illustrate the provision of ecosystem services, their economic value, costs of operation and net impact on livelihoods and wellbeing.

Conducting a global assessment of the costs and benefit of expanding MPA coverage requires a number of assumptions and comes with numerous caveats and limitations. To ensure the transparency of how this assessment is conducted, we state the key assumptions, caveats and limitations of the analysis in the relevant sections of the report. In addition, section 6 provides a discussion of the main caveats and limitations.

⁵ It is noted the CBD Aichi Target 11 does not specify that 10% of coastal and marine area should be conserved specifically as no-take MPAs. The examination of only no-take MPAs in this study is a necessary restriction of the analysis. It is also defensible given that recent major global recommendations for no-take MPAs exceeding 10% coverage have been made (e.g. the Promise of Sydney).

3 Methods

The general methodological framework for the analysis follows that of Balmford *et al.* (2011), Bateman *et al.* (2011), Hussain *et al.* (2011) and Brander *et al.* (2012b). In particular it incorporates several critical insights from the environmental economics literature by: contrasting counterfactual scenarios that differ solely in whether they include policy interventions; identifying non-overlapping ecosystem services; modelling spatially-explicit variation in the values of ecosystem services; and comparing the benefits of conservation policies with the costs. The methodological framework is represented in Figure 1. The specific methodologies used to operationalize this assessment framework are described in the following sections.

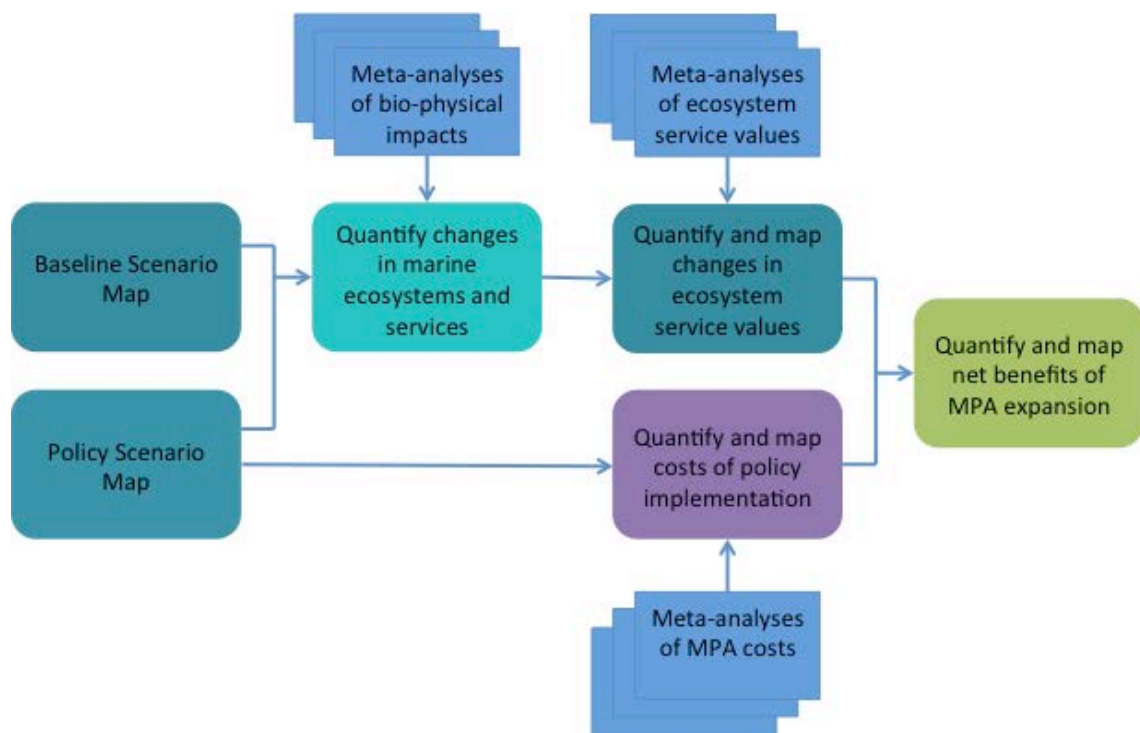


Figure 1 Methodological framework for assessing the net benefits of expanding marine protected areas. Adapted from Figure 2, Balmford *et al.* (2011); and Figure 2, Hussain *et al.* (2011).

3.1 Comparison of scenarios

To answer the main question posed by this study (i.e., what are the net benefits of protecting marine habitats through expanding the coverage of no-take MPAs?) we need to develop descriptions of the future (scenarios) for what an expansion of MPA coverage might look like, and assess the net benefits of these 'exploratory scenarios' relative to a baseline scenario of no additional expansion of MPAs. Figure 2 provides a conceptual representation of the comparison of scenarios. The upper panel represents the change in the proportion of total marine area that is designated as MPA over time. The lower panel represents the resulting change in the value of marine ecosystem services over time. The net benefits of expanding MPA coverage is assessed as the change in the value of marine ecosystem services relative to the baseline scenario

(represented by the shaded area in the lower panel) minus the costs of expanding MPAs. It is important to note that Figure 2 is only a conceptual representation of how scenarios are compared. The slopes of the lines in the lower panel do not represent assumptions underlying the analysis. The extent to which the value of marine ecosystem services changes under the baseline and is affected by expanding MPA coverage is quantitatively modelled in the analysis.

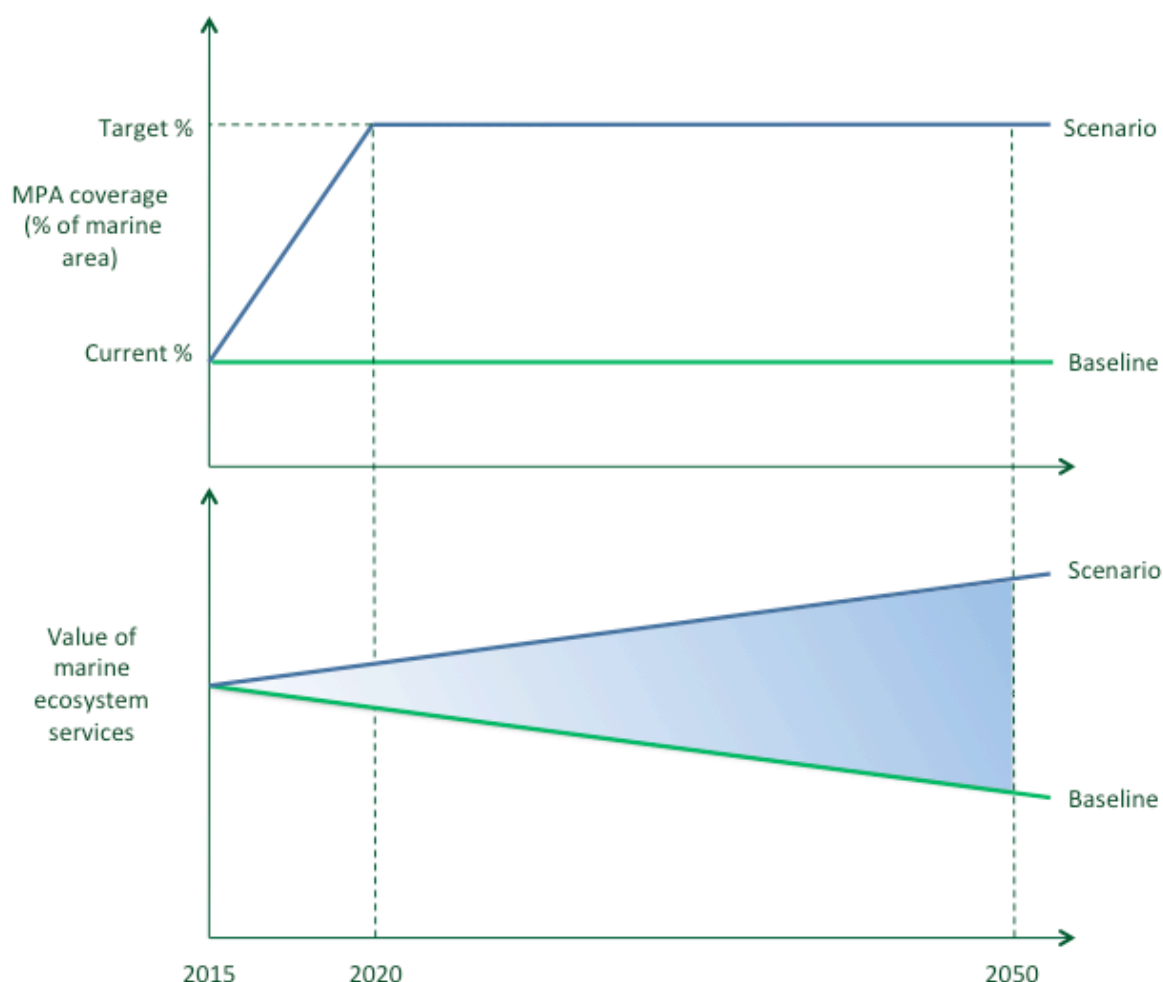


Figure 2 Conceptual representation of the comparison between baseline and exploratory scenarios. The added value of expanding MPA coverage is represented by the shaded area in the lower panel.

3.2 Scenario development

The analysis undertaken in this study requires a description of the location and extent of MPAs in the absence of any additional expansion of MPA coverage. In other words, we require a baseline scenario.⁶ This is necessary because the analysis involves making a comparison between the value of marine ecosystem services under exploratory

⁶ Alternative terms for a baseline scenario are 'business-as-usual', 'counter-factual' or 'policy off' scenarios. Note that a baseline scenario does not necessarily imply a continuation of the current situation into the future but provides a dynamic description of what the future will look like without the intervention that is the focus of the analysis.

‘policy’ scenarios of expanding MPA coverage versus the value if no additional action is taken. The baseline extent of MPAs is held constant at the current coverage (3.4%). In addition to a description of the extent and location of MPAs, the baseline scenario also needs to provide information on a number of important factors that are changing over time and are likely to have considerable bearing on the benefits and costs of expanding MPAs. The baseline scenario describes the future as it is likely to develop following current trends, threats and pressures. Such factors include population, income, land based pollution, sedimentation, infrastructure development, climate change and ocean acidification. Regarding the baseline impacts of climate change on marine ecosystems, we make use of the spatially explicit threat levels modelled in the Reefs at Risk Revisited study (Burke *et al.*, 2011). These factors change over the time horizon of the analysis (2015-2050) but are held constant across all scenarios (i.e. the analysis is focused on changes in MPA coverage only).

The questions to be answered in developing specific exploratory scenarios are: What extent of marine area should be designated as MPA? What criteria can be used to locate new MPAs? Accordingly, the scenarios for MPA expansion are developed along these two dimensions:

1. The proportion of marine area designated as no-take MPA. We explore two alternative extents of areal coverage: 10% and 30%. These area targets were selected to loosely correspond with those of the CBD Aichi Target 11 and the upper limit of the Durban Action Plan. It is not the intention, however, that our scenarios model all aspects of the CBD or Durban targets.
2. The spatial location of MPAs. Two criteria are used to determine the spatial location of MPAs: 1) marine biodiversity; 2) exposure of marine ecosystems to human impacts. Alternative combinations of these two criteria allow us to assess the relative net benefits of targeting MPAs for preservation or restoration. MPAs may be effective at achieving both. The possible combinations of these two criteria are given in Table 1. In targeting locations that are characterised by high biodiversity and high human impact, the protection arguably serves to mitigate damage (“Protect to Mitigate”). Alternatively, targeting areas with high biodiversity and low human impact provides protection to intact ecosystems from potential future human impact (“Protect to Preserve”). Targeting areas with low biodiversity and low human impact identifies locations for which it is likely to be easier to expand MPAs (“Easy to Expand”). The final combination of areas with low biodiversity and high human impact do not represent plausible locations for expanding MPAs and are not considered further. In addition, we impose the requirements that the target percentages for MPA coverage are applied to each habitat type, national exclusive economic zone (EEZ) and area beyond national jurisdiction (ABNJ). It is important to note that we do not otherwise attempt to ensure ecological representativeness in the location of MPAs.

Table 1 Exploratory scenarios for MPA expansion derived from alternative combinations of biodiversity and human impact criteria

	Low Human Impact	High Human Impact
Low Biodiversity	Easy to Expand -E2E- <i>Low Biodiversity Areas with Low Human Impact</i>	–
High Biodiversity	Protect to Preserve -P2P- <i>High Biodiversity Areas away from Human Impact. Protection preserves the area from potential future impact</i>	Protect to Mitigate -P2M- <i>High Biodiversity Areas under High Human Impact: Protection is mitigating the Impact</i>

The combination of all variants of the two dimensions (proportion of marine area and target locations) gives six exploratory scenarios that are subsequently mapped. It is recommended to think of the set of scenarios as explorative; they pose the question “what if MPAs were expanded in this way?”

Global data on species biodiversity were obtained from www.aquamaps.org (accessed on 19 Aug. 2014) and data on human impact on marine ecosystems were obtained from Halpern *et al.* (2008). These data, represented in Figures 3 and 4 respectively, are used to create MPA allocation priority maps for each scenario. The priority maps were then fed into a model together with jurisdictional data (EEZs and the ABNJs were further divided per FAO fishing area). Due to issues of data quality, no areas beyond 70 degrees North or South are included in the analysis.

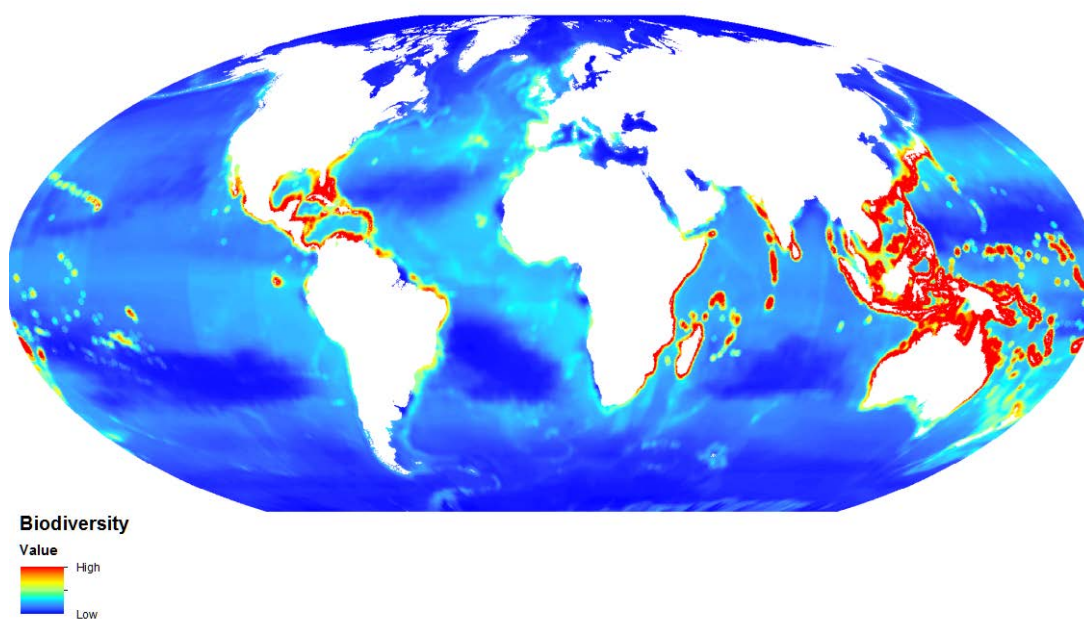


Figure 3 Global all species biodiversity map. Source: www.aquamaps.org (accessed on 19 Aug. 2014)

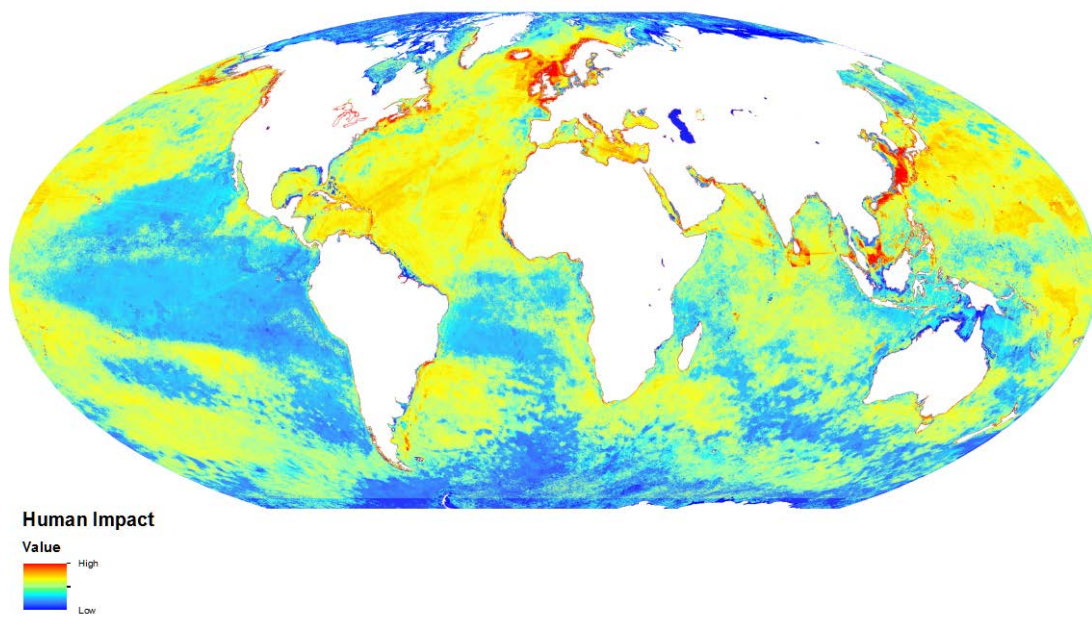


Figure 4 Global cumulative human impact map. Source: Halpern et al. (2008)

A schematic representation of the model is shown in Figure 5. Existing MPAs (source: UNEP-WCMC) are retained in the scenario maps. If a country currently meets the targeted coverage of MPA as a proportion of its EEZ, no reallocation of MPAs takes place and existing MPAs are represented in the scenario maps. New MPAs are allocated using the following allocation rules:

1. Each Key Habitat has the same extent of protection in terms of proportion of area
2. Each EEZ has the same extent of protection in terms of proportion of area
3. Each ABNJ planning unit has the same extent of protection in terms of proportion of area

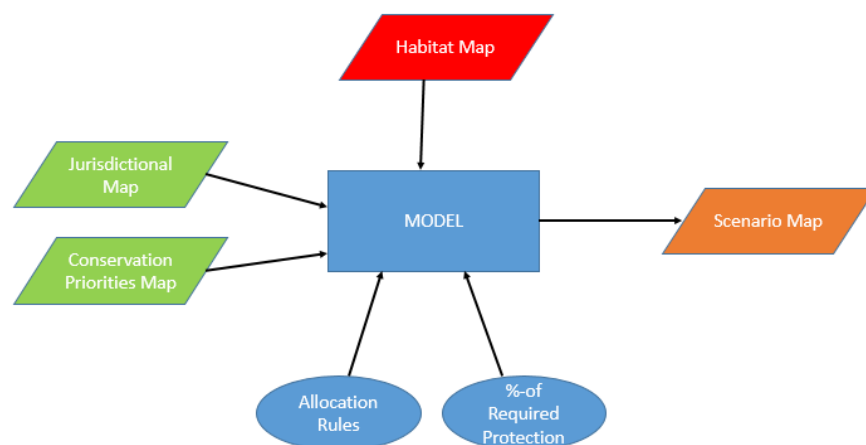


Figure 5 Schematic diagram of scenario mapping model

A detailed explanation of the criteria, data and models used in developing the exploratory scenarios for expanding MPA coverage is provided in Appendix A.

3.3 GIS analysis of marine habitats and MPAs

Taking the mapped scenarios for MPA expansion as a starting point, GIS analysis is used to derive spatial data on:

1. The characteristics of individual MPAs with which to compute establishment, operation and opportunity costs; and employment. The required characteristics are defined in the respective cost and employment functions (see sections 4.2 and 4.5).
2. The characteristics of marine habitats with which to estimate MPA effects on ecosystem services and values. The specific characteristics are defined in the respective value functions for each marine habitat (see section 4.3)

Software

The variables required for the analysis of marine protected areas (MPA), corals, mangroves and other coastal wetlands are generated using primarily *R* version 3.1.1 and the most current versions of the *R* packages *raster*, *rgeos* and *sp*. ArcGIS (version 10.2.1) is used for basic data preparation where *R* did not provide the required functionality.

Data sources

The input data for the analyses comprise both raster data and shapefiles.

The shapefiles of MPAs under various scenarios are generated in the scenario development (see Section 3.2). Coastal wetlands were extracted from the Global Lakes and Wetlands Database Level 3 (Lehner & Döll 2004). The shapefile with global coral reefs was developed for the Reefs at Risk Revisited project (Burke *et al.* 2011). The Reefs at Risk Revisited data also include a raster with projections for threat levels (an indicator composed of bleaching, human impacts, etc.). The shapefile of global mangroves used in this project was developed by Giri *et al.* (2011).

The variables for the required value transfer functions (see sections 4.2 and 4.3 for the specific cost and benefit functions) came from the following sources. The rasters with (actual) net primary production (NPP) and human appropriation of net primary production (HANPP) were developed by Haberl *et al.* (2007). For population density (adjusted to UN statistics), the raster developed by CIESIN & CIAT (2005) was used. Data on roads and ports were downloaded from the open-access data hub Natural Earth Data at the highest available resolution.⁷

Data processing

For each variable and each type of marine or coastal ecosystem, a separate *R* script was developed with the appropriate functions. The functions are available in the *R* packages listed above and provide basic functionality for spatial analysis (e.g., *gLength*, *gArea*, *cellStats*). These scripts are available upon request.

Generally, the contextual variables were calculated in an equal-area projection unless preservation of distance was important, in which case data were reprojected in an equal-distance projection. Regarding the contextual variables in the value transfer functions for corals, mangroves and coastal wetlands (see section 4.3), a 50 kilometre radius was drawn around site centroids for overlays with the required raster data or shapefiles.

⁷ <http://www.naturalearthdata.com>

The mangroves shapefile is highly detailed, which makes processing cumbersome. Therefore, the decision was made to consider only mangrove sites larger than 5 hectares. The ecosystem “coastal wetlands” in the Global Lakes and Wetlands Database contains mangroves, so an overlay with the mangrove map was made to correct the area of coastal wetlands. The value transfer functions of both coastal wetlands and mangroves consider the area of coastal wetlands and mangroves (respectively) in a 50 kilometre radius, and here a correction was made to consider the area of each individual site.

3.4 Literature review and meta-analysis

In order to assess the net economic benefits of MPA expansion we require a quantitative understanding of the:

- Bio-physical impacts of MPAs on the marine environment;
- Associated change in the provision of ecosystem services;
- Economic value of marine ecosystem services; and
- Establishment, operating and opportunity costs of MPAs.

We obtain these quantitative relationships through extensive literature reviews and, where feasible, meta-analyses of the relevant literature. The scope of this study and global nature of the analysis did not allow for primary data collection and analysis of these quantitative relationships.

Meta-analysis is a method of synthesizing the results of multiple studies that examine the same phenomenon, through the identification of a common effect, which is then ‘explained’ using regression techniques in a meta-regression model (Stanley, 2001). Meta-analysis was first proposed as a research synthesis method by Glass (1976) and has since been developed and applied in many fields of research, not least in the area of environmental economics (Nelson and Kennedy, 2009). Given the large literatures that exist on the bio-physical impacts of MPAs, the impact of MPAs on the provision of ecosystem services, the economic value of marine ecosystem services, and the costs of establishing and operating MPAs, there is a need for research synthesis techniques, and in particular statistical meta-analysis, to aggregate information and insights (Stanley, 2001; Smith and Pattanayak, 2002; Bateman and Jones, 2003). In addition to identifying consensus in results across studies, meta-analysis is also of interest as a means of transferring parameter values from studied sites to new policy sites (Rosenberger and Phipps 2007). It is for this purpose that we use the results of meta-analyses in this report.

The literature reviews on the impacts of MPAs on bio-physical characteristics and marine ecosystem services are described in detail in Appendices 2 and 3 respectively. The selection and application of specific quantitative relationships obtained from the literature are detailed in the relevant results sections

3.5 Value transfer

Value transfer involves estimating the value of ecosystem services through the use of value data and information from other similar ecosystems and populations of beneficiaries (Navrud and Ready, 2007; Brander, 2013). It involves transferring the results of existing primary valuation studies for other ecosystems (“study sites”) to ecosystems that are of current policy interest (“policy sites”). This procedure is also known as benefit transfer but since the values being transferred may also be estimates of costs or damages, the term value transfer is arguably more appropriate (Brouwer,

2000). In this study we transfer existing information on ecosystem services values, MPA costs and employment, and quantified impacts of MPAs on marine ecosystems and ecosystem services to inform our global analysis of the costs and benefits of expanding MPA coverage.

The use of value transfer to provide information for decision making has a number of advantages over conducting primary research to estimate ecosystem values. From a practical point of view it is generally less expensive and time consuming than conducting primary research. Value transfer can also be applied on a scale that would be unfeasible for primary research in terms of valuing large numbers of sites across multiple countries. Value transfer also has the methodological attraction of providing consistency in the estimation of values across policy sites (Rosenberger and Stanley, 2006).

In this study we largely conduct value transfers using functions obtained from meta-analyses. The meta-analytic function transfer technique uses a value function estimated from the results of multiple primary studies representing multiple study sites in conjunction with information on parameter values for the policy sites to calculate the value of an ecosystem service at the policy site (e.g. individual marine ecosystem). This method is represented in Figure 6. A value function is an equation that relates the value of an ecosystem service to the characteristics of the ecosystem and the beneficiaries of the ecosystem service. Since a meta-analytic value function is estimated from the results of multiple studies it is able to represent and control for greater spatial variation in the characteristics of ecosystems, beneficiaries and other contextual characteristics that cannot be generated from a single primary valuation study.

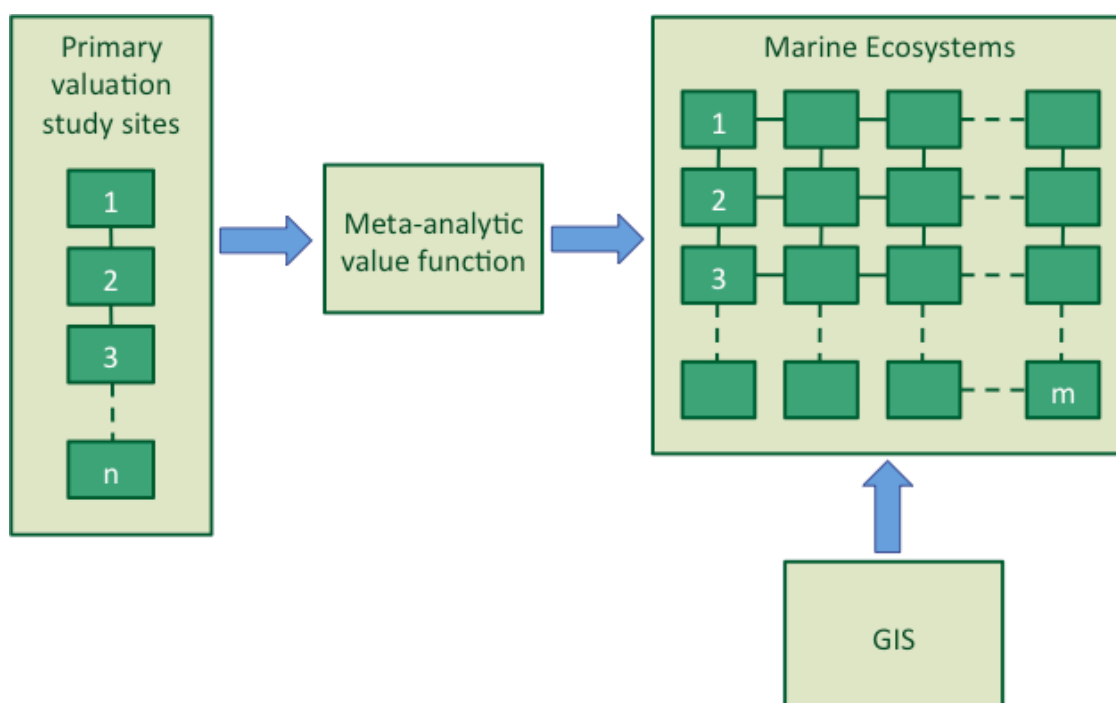


Figure 6 Meta-analytic value transfer method

A general specification of a value function is given equation 1. It states the relationship between value (e.g. value of ecosystem services provided by coral reefs) and a set of explanatory variables that describe the site (e.g. area of the coral reef), context (e.g.

abundance of other ecosystems in the vicinity) and socio-economic characteristics (e.g. income levels and population in the vicinity). The β s are empirically estimated coefficients that describe the relationships between values and the explanatory variables. Using information on the relevant explanatory variables for each policy site obtained using GIS (see Section 3.3.), we apply value functions to predict values for each marine ecosystem. The value functions that are identified through the literature reviews and used in our analysis are described in full in Section 4.

$$\text{Value} = \beta_S * \text{Site Characteristics} + \beta_C * \text{Context Characteristics} + \beta_E * \text{Socio-Economic Characteristics} \quad (\text{Eq. 1})$$

An important consideration in estimating the value of changes to a biome across a large geographic area, such as we do in this study, is that changes in the stock of the resource may affect the unit values of each individual ecosystem. Localised changes in the extent of any individual ecosystem may be adequately valued in isolation from the rest of the stock of the resource, which is implicitly assumed to be constant. When valuing simultaneous changes in multiple ecosystem sites within a region (e.g., global expansion of MPAs to 10-30% coverage of total marine area), it is arguably not sufficient to estimate the value of individual ecosystem sites and aggregate them without accounting for the changes that are occurring across the stock of the resource. We therefore follow the method proposed by Brander *et al.* (2012b) to include spatial information in the meta-analytic value functions on the abundance of marine ecosystems in the broader surroundings of each study and policy site. This variable is used to capture the effect of changes in the availability of substitute or complementary ecosystems in the vicinity of each ecosystem site. In addition, a number of other characteristics of each ecosystem site derived from spatial data are included in the analyses as potential determinants of ecosystem value.

Value transfer limitations and sources of uncertainty

Using value transfer methods is arguably the only viable means of estimating ecosystem service values at a global scale but it is important to note the limitations and potential inaccuracies involved. Ecosystem service values estimated using value transfer methods may be inaccurate for a number of reasons (Rosenberger and Stanley, 2006). In other words, transferred values may differ significantly from the actual values of the ecosystem services at the policy site. The main sources of uncertainty in the values estimated using value transfer are (Brander, 2013):

1. Primary value estimates used in value transfer are themselves uncertain. Inaccuracies in primary valuation estimates may result from weak methodologies, unreliable data, analyst errors, and the whole range of biases and inaccuracies associated with primary valuation methods.
2. The available stock of information on ecosystem service values may be unrepresentative due to the processes through which primary valuation study sites are selected and results are disseminated, which can be biased towards certain locations, services, methods and findings (Hoehn, 2006; Rosenberger and Johnston 2009).
3. The number of reliable primary valuation results may be limited, particularly for certain services, ecosystems and regions. As the number and breadth of high quality primary valuations increases, the scope for reliable value transfer also increases. For some ecosystems, ecosystem services and regions there are now

many good quality value estimates available whereas for others there are still relatively few.

4. The process of transferring study site values to policy sites can also potentially result in inaccurate value estimates (Rosenberger and Phipps 2007). So-called 'generalisation error' occurs when values for study sites are transferred to policy sites that are different without fully accounting for those differences. Such differences may be in terms of beneficiary characteristics (income, culture, demographics, education etc.) or biophysical characteristics (quantity and/or quality of the ecosystem service, availability of substitutes, accessibility etc.). The availability of study sites that are closely similar to policy site and/or the value transfer methods used to control for differences will determine the magnitude of generalisation error.
5. There may also be a temporal source of generalisation error since preferences and values for ecosystem services may not remain constant over time. A value function that is able to predict current values well may not perform as well in predicting future values.

Steps in meta-analytic value transfers

In this section we provide a general explanation of the meta-analytic value transfer methodologies that are used to estimate the benefits and costs of expanding MPA coverage. The estimation of carbon sequestration benefits and opportunity costs to commercial fisheries apply different value transfer methods and are described separately below. The parameter values, value functions, and data sources that are used to estimate each benefit and cost are provided in the results section (Section 4).

The methodology used to estimate the change in value of marine ecosystem services following expansion of MPA coverage takes the following steps:

1. Conduct a literature review to obtain existing meta-analytic value functions that relate ecosystem service value to the characteristics of the ecosystem and its surroundings.
2. Using GIS, develop global databases of marine ecosystems containing information on the variables included in the value functions obtained in step 1.
3. Using the databases developed in step 2, compute baseline change in the spatial extent of each marine ecosystem using estimates of future rates of loss obtained from the literature review. Where possible, baseline change is spatially variable to reflect variation in pressures on ecosystems.
4. Compute the difference in the spatial extent of each ecosystem between exploratory scenarios and the baseline. Differences in spatial extent of ecosystems resulting from protection are obtained from the literature review of bio-physical effects of MPAs (see Appendix B). This gives us the additional area of each marine ecosystem under each exploratory scenario that would not exist under the baseline.
5. Input the data generated in steps 2-4 into the value functions obtained in step 1 to estimate the value of changes in marine ecosystem services under each exploratory scenario relative to the baseline scenario. It is important to note that the scale at which this analysis is conducted is at the level of individual marine ecosystem sites or patches (e.g. individual coral reefs or mangrove forests). This scale of analysis allows the estimation of values that are specific to the characteristics and context of each individual marine ecosystem.

The methodology used to estimate the establishment costs, operation costs and employment effects of expanded MPA coverage follows a similar but slightly different set of steps:

1. Conduct a literature review to obtain existing meta-analytic cost functions that relate MPA cost to the characteristics of the MPA.
2. Using GIS, develop global databases of MPAs under each exploratory scenario containing information on the variables included in the cost functions obtained in step 1.
3. Input the data generated in step 2 into the cost functions obtained in step 1 to estimate the cost of expanding MPA coverage under each exploratory scenario relative to the baseline scenario. It is important to note that the scale at which this analysis is conducted is at the level of individual (geographically separate) MPAs. This scale of analysis allows the estimation of costs and employment that are specific to the characteristics and context of each individual MPA.

Steps in value transfer for carbon storage in mangroves

The method used for estimating the value of additional carbon stored in coastal and marine ecosystems does not employ a meta-analytic value function. The reason for treating carbon sequestration differently from other ecosystem services is that, as a global pollutant, the economic value of carbon emissions does not vary spatially, whereas the economic values of other ecosystem services are highly spatially variable, which requires the use of value transfer methods that reflect this.⁸ It is generally the case that the values of other ecosystem services are scale-dependent whereas this is not the case for carbon sequestration. The unit value of carbon sequestration is independent of scale because each avoided tonne of CO₂ emission in a given time period performs the same level of climate stabilization service (Murray *et al.*, 2011).

Mangroves, wetlands, coral reefs, seagrasses, and algae floating at sea all remove carbon dioxide from the atmosphere and store it in their fibres, in the soil, and in the ocean substrate. The amount of carbon that is captured from the atmosphere by different organisms can be quantified in terms of a rate of sequestration. If a tree or plant is destroyed, the carbon stored in the plant's cells is released as the biomass decays or burns. Carbon stored in the soil/substrate may be released over time if left un-vegetated, or released quickly if the substrate is disturbed. Both the rate at which carbon is added to biomass/substrate and any release of stored carbon are important for estimating the total value of avoided ecosystem loss. Together they represent the net carbon sequestered from the atmosphere, or change in the stock of stored carbon, in a given time period. The net amount of carbon sequestered by an ecosystem in a given time period is the sum of the rate of sequestration of each species ($r_{s,t}$) and the release of stored carbon ($q_{s,t}$)

$$\text{Carbon Sequestration}_t = \sum (r_{s,t} - q_{s,t}) \quad (\text{Eq. 2})$$

The subscript s refers to the species; the subscript t refers to the length of time analysed, usually one year. Data on the rates of carbon sequestration by different ecosystems and the extent of those ecosystems can be used to estimate annual

⁸ The values of marine ecosystem services are unlikely to be the same for each individual ecosystem site and may be expected to vary with the size of the ecosystem, availability of substitute ecosystems nearby, and the number of beneficiaries that live nearby and make use of the ecosystem services. The analysis explicitly attempts to account for such (spatial) variation in the value of ecosystem services produced by each ecosystem site.

quantities of carbon sequestration; data on the quantity of stored carbon in different ecosystems and reductions in extent of those ecosystems can be used to estimate the annual quantity of released carbon.

The net quantity of carbon sequestered, multiplied by the value per tonne of carbon is an estimate of the annual value of carbon sequestration by an ecosystem, as represented by equation 3 below.

$$\text{Value Carbon Sequestration}_t = \sum(r_{s,t} - q_{s,t}) * \text{Value per tonne carbon} \quad (\text{Eq. 3})$$

By convention, quantities of carbon are expressed in terms of tonnes of CO₂-equivalent in order to allow comparison with other greenhouse gases. In our analysis, quantities and values of carbon are expressed in tCO₂. The conversion rate between carbon and CO₂ is 1 tC = 3.67 t CO₂.

The value of avoided carbon emissions and additional sequestration by mangroves is estimated using the methods and parameters described in Murray *et al.* (2011) and Pendleton *et al.* (2012). The methodology used for estimating the quantity and value of additional carbon stored in marine and coastal ecosystems due to expansion of MPA coverage take the follow steps:

1. Obtain data on the current spatial extent of marine ecosystems (see Section 3.3)
2. Compute the areal extent of marine ecosystems in each year of the analysis under the baseline scenario using loss rates obtained from the literature.
3. Compute the avoided loss in areal extent of marine ecosystems under each exploratory scenario relative to the baseline assuming that protection prevents loss but does not result in recovery. This involves subtracting the area of mangrove under the baseline scenario in each year from the area of mangrove in the initial year of the analysis (2015).
4. Compute the additional carbon sequestration under each exploratory scenario relative to the baseline by multiplying the cumulative avoided loss of ecosystem area (from step 3) by estimates of sequestration per unit area obtained from the literature. For mangroves we use a rate of 6.3 tCO₂/ha/year from Pendleton *et al.* (2012).
5. Compute the avoided release of carbon stored in biomass and substrate by multiplying the avoided loss of ecosystem area by estimated rates of release. The rates at which stored carbon is released following ecosystem loss is different for biomass and substrate carbon and depends on the extent of disturbance to substrate. For mangroves, Murray *et al.*, (2011) assume that 75% of biomass carbon is released immediately and that the remaining 25% decays with a half-life of 15 years (i.e. a further 12.5% is released within 15 years, a further 6.25% is released within 15 years after that, etc.). They further assume that mangrove soil organic carbon has a half-life of 7.5 years (i.e. 50% of the stored carbon is released in the first 7.5 years, 25% in the following 7.5 years, etc.).
6. Compute the total additional carbon stored in each year of the analysis (i.e. sum estimates from steps 5 and 6 for each year).
7. Compute the value of additional carbon stored in each year of the analysis by multiplying the estimated total quantity (from step 6) by the value per tonne CO₂ for each year. The relevant value per tonne of CO₂ is the social cost of carbon (SCC), which is the monetary value of damages caused by emitting one more tonne

of CO₂ in a given year (Pearce, 2003).⁹ The SCC therefore also represents the value of damages avoided for a small reduction in emissions, in other words, the benefit of a reduction in atmospheric CO₂ in a given year. The SCC increases over time due to the increasing marginal damage caused by additional tonnes of CO₂ in the atmosphere. In our analysis we use the US Interagency Working Group series of SCC estimates for the period 2010-2050 (Interagency Working Group, 2013).¹⁰

Steps in valuation of fisheries impacts

The calculation of the impact of MPA designation on commercial fisheries involves multiple steps (represented in Figure 7) that gather several data sources:

1. FAO data for both values and quantity of fisheries exports (The State of World Fisheries and Aquaculture – 2012)¹¹ was used to estimate a US\$ per tonne value for marine capture fisheries production. This calculation gives a proxy estimate of the unit value of production.
2. The total value of fisheries is then divided by the global ocean area to get an average value of fisheries production per km².
3. For each country the area of existing MPAs is subtracted from the estimated total MPA area for each of the scenarios being evaluated. This gives the change in MPA area (km²) by country under each MPA scenario.
4. The change in MPA area and value per km² are combined to estimate the value of reduced fisheries production under each scenario.
5. It is assumed in each of the scenarios that the additional MPAs are no-take areas, and there are beneficial spillovers in terms higher fish stocks in areas not designated as MPAs. More accurately we assume that current capture fisheries are not sustainable and will decline over time, the spillover impact of MPA designation reduces the rate of fisheries decline in the non-MPA areas. This reduction in the rate of decline is higher for the 30% MPA scenarios compared to the 10% MPA scenarios.

⁹ The SCC is intended to be a comprehensive estimate of climate change damages but due to current limitations in the integrated assessment models and data used to estimate SCC, it does not include all important damages and is likely to under-estimate the full damages from CO₂ emissions.

¹⁰ An alternative value per tonne CO₂ that is commonly used in the appraisal of emissions reductions is the observed price in carbon markets. The problem with this approach is that prices in carbon markets are largely artefacts of the set up and regulation of the market and do not reflect the benefits of carbon sequestration.

¹¹ <http://www.fao.org/docrep/016/i2727e/i2727e.pdf>

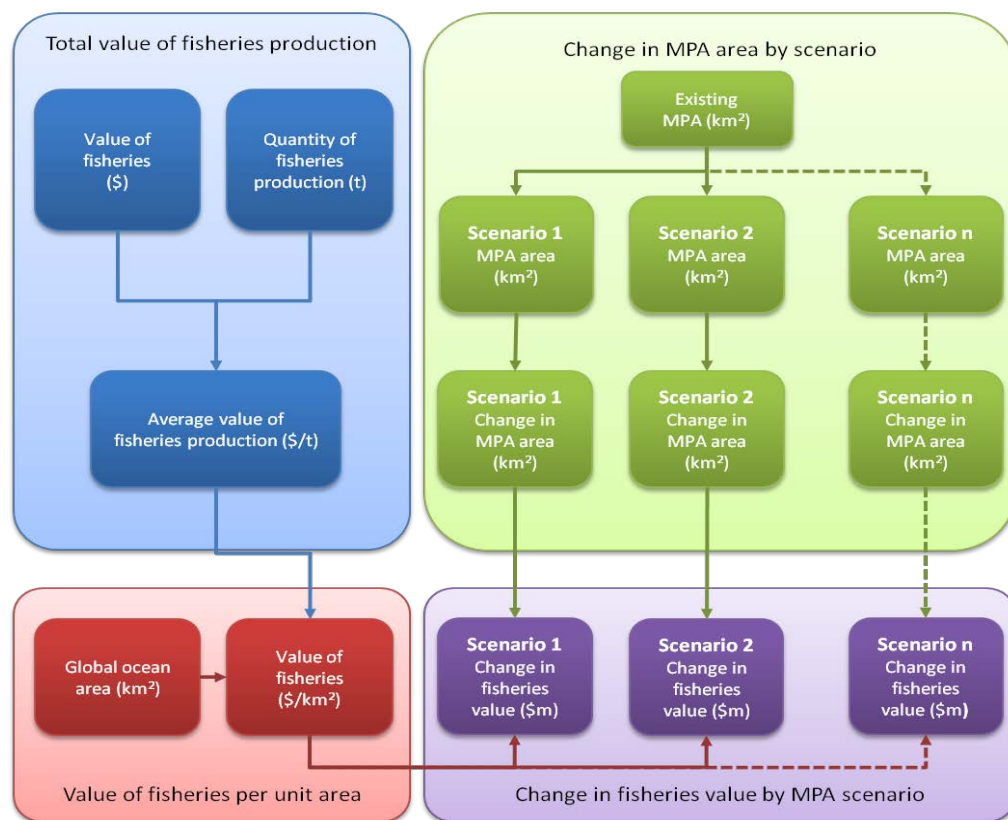


Figure 7 Methodology for assessing MPA effects on commercial fisheries

3.6 Cost-benefit analysis

Cost benefit analysis (CBA) is a method in which the societal costs and benefits of alternative options or scenarios are expressed and compared in monetary terms. CBA provides an indication of how much a prospective investment contributes to social welfare by calculating the extent to which the benefits of the project exceed the costs.

The methodology for the CBA takes the following steps:

1. Quantify negative and positive effects (costs and benefits) of expanding MPAs in monetary units (see section 3.5). This gives a time-series of future values for each cost and benefit over the time horizon of the analysis. The time horizon is the period over which effects are assessed. The time horizon of our analysis is 2015-2050, which provides a sufficiently long period over which the benefits of MPAs can be realised.
2. Convert cost and benefits that are expressed in the price levels of different years to a common price level.¹² We use GDP deflators from the World Bank World Development Indicators to convert all values to 2012 price levels.¹³

¹² Value estimates may be reported at the general price level for a particular year, usually the year in which the study was conducted. For example, a valuation study conducted in 2005 is likely to report values in the price level in that year. Inflation, however, causes general price levels to rise over time so that any given amount of money is worth less, in terms of the goods and services that it can purchase, over time.

¹³ <http://data.worldbank.org/data-catalog/world-development-indicators>

3. Convert future values of costs and benefits to present values (2015) reflecting society's time preference. This involves discounting the value of costs and benefits that occur in future years. In this analysis we use a discount rate of 3%, which is in line with similar global assessments (Hussain *et al.*, 2011). The formula for discounting or calculating present values is:

$$PV = FV / (1+r)^n \quad (\text{Eq. 4})$$

where:

PV = present value

FV = future value

r = discount rate

4. Compute total present values across each cost and benefit category by summing each time-series of costs and benefits.
5. Compute total present value costs and benefits by summing across all costs categories and benefit categories.
6. Compute the net present value (NPV) of each exploratory scenario by subtracting the sum of present value costs from the sum of present value benefits. A positive NPV indicates that scenario represents an improvement social welfare. The NPV formula is:

$$NPV = \sum_{t=1, \dots, T} (B_t - C_t) / (1+r)^t \quad (\text{Eq. 5})$$

where:

B = benefits

C = costs

r = discount rate

t = years from base year

T = time horizon

7. Compute the benefit cost ratio (BCR) of each exploratory scenario as the sum of discounted benefits and the sum of discounted costs. The BCR shows the extent to which benefits exceed costs under each scenario. A BCR greater than 1 indicates that the benefits of a scenario exceed the costs. BCR formula:

$$BCR = \sum_{t=1, \dots, T} B_t / (1+r)^t / \sum_{t=1, \dots, T} C_t / (1+r)^t \quad (\text{Eq. 6})$$

where:

B = benefits

C = costs

r = discount rate

t = years from base year

T = time horizon

8. Compute the internal rate of return (IRR) of each exploratory scenario as the discount rate at which a scenario's NPV becomes zero. If the IRR exceeds the discount rate used in the analysis (i.e. 3%), the scenario can be considered economically worthwhile.

Ideally the CBA would include all relevant costs and benefits associated with the expansion of MPAs. These are listed in Table 2. Due to limitations on the available data and knowledge of how MPAs affect ecosystems and the provision of ecosystem services, we are currently only able to include a sub-set of costs and benefits in the analysis presented in the section 4. The cost and benefit categories that we are able to include in the CBA are indicated in Table 2.

Table 2 Costs and benefits included in the analysis

Cost category	Included in CBA
MPA establishment costs	Yes
MPA operational costs	Yes
Opportunity costs to commercial fisheries	Yes
Opportunity costs to shipping	No
Opportunity costs to mineral extraction	No
Opportunity costs to power generation	No
Benefit category	
Coastal wetland ecosystem services	Yes
Coral reef ecosystem services	Yes
Mangrove ecosystem services (including carbon storage)	Yes
Seagrass ecosystem services	No
Sea mount ecosystem services	No
Ocean absorption of carbon dioxide	No
Bio-prospecting	No
Positive spill-over effects to fisheries	No
Non-use values for marine biodiversity	No

4 Results

4.1 Scenarios for MPA expansion

The mapped scenarios for the expansion of MPA coverage, following the development process and allocation rules described in section 3.2, are not presented in this report due to concerns that they could be misinterpreted as spatially explicit recommendations for the siting of MPAs. The development and siting of specific MPAs and networks of course requires a rigorous process of research, consultation and assessment that reflects multiple factors relevant to each case. The scenarios developed in this study do not replicate that process and should only be used for the purpose in which they are intended: the exploratory assessment of the potential global net benefits of expanding MPA coverage.

4.2 Costs

There are two broad categories of cost associated with the creation and management of marine protected areas: those that are incurred by the government or implementing agency in establishing and operating the MPA, and those that are incurred by industry and coastal communities in the form of compliance and opportunity costs. Consideration of both of these types of costs is an important part of assessing the economic feasibility of establishing, expanding, or altering MPAs (Grafton *et al.*, 2011; McCrea-Strub *et al.*, 2011).

The governmental, or institutional, costs comprise the costs of establishing a specific MPA and the recurrent costs associated with the actual operation and management of the MPA following its designation. The establishment costs include all costs incurred up to and including the designation of the MPA and the initiation of its management (McCrea-Strub *et al.*, 2011), whereas all costs incurred subsequently can be classified as recurrent operational costs.

Establishment costs of expanding MPAs

Establishment costs have been less frequently estimated than recurrent operational costs. One of the challenges in estimating establishment costs is that until a single management entity is created to administer an MPA, cost and budgetary data is spread across a range of government entities, making this data hard to collate (McCrea-Strub *et al.*, 2011). Studies that have examined MPA establishment costs indicate that these costs are spatially heterogeneous at a fine scale (Richardson *et al.*, 2006), and that they are subject to economies of scale such that larger networks tend to have lower establishment costs per unit area than do smaller networks (McCrea-Strub *et al.*, 2011). Furthermore, the location of MPAs are an important determinant of establishment costs because this has a direct effect on the cost of consultations, impact assessments, and the designations themselves. Based on a survey of the literature (Ban *et al.*, 2011; Ban and Klein, 2009; Cook and Heinen, 2005; Gleason *et al.*, 2013; Hunt, 2013; Leisher *et al.*, 2012; McCrea-Strub *et al.*, 2011; Vincent *et al.*, 2004), establishment costs may be further broken down into approximately eight different types of cost: benchmark ecological and stakeholder assessment; build key partnerships; communication activities; planning and design; initiation of management structures; initiation of monitoring, compliance and enforcement; demarcation of MPA boundary; initiation of necessary off-reserve management and legislation (Baulcomb, 2013). For the assessment of the establishment costs of expanding MPA coverage we

make use of Model D from McCrea-Strub *et al.* (2011), which relates the establishment cost per km² to the area of the MPA (see Table 3). This model describes a negative empirical relationship between cost per km² and the size of an MPA, suggesting that there are economies of scale in increasing the size of MPA. It may be the case, however, that larger MPAs also have lower levels of management and effectiveness.

Data on MPA areas under each scenario are fed into this cost function to estimate the establishment costs for each MPA. We assume that these costs are incurred over the period 2015-2020 in equal annual instalments. The estimated costs are converted from 2005 values to 2013 values using a GDP deflator conversion factor from the World Bank World Development Indicators (2015). Table 4 presents the total MPA establishment costs for each scenario. The costs of establishing MPAs increase with the extent of MPA coverage but not at a linear rate. There are substantial economies of scale, i.e. the cost per unit area decreases as the area of an MPA increase. The P2M30 scenario stands out as having considerably higher establishment costs than the other scenarios. This is explained by the size distribution of MPAs under this scenario. There are no very large (relatively low cost) MPAs under this scenario.

Table 3 MPA establishment cost function. Source: Table 4, McCrea-Strub et al. (2011)

Variable	Units	Coefficient
Establishment cost	2005 USD/km ² ; log ₁₀	
Intercept		4.66
MPA area	km ² ; log ₁₀	-0.48

Table 4 MPA establishment costs (USD; billions; 2013 price level; present values using a discount rate of 3%)

Scenario	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Cost (billions)	10.451	10.751	10.007	21.870	10.441	12.180

Operational costs of expanding MPAs

Operational costs have received more attention and existing research indicates that operational costs also experience economies of scale (Balmford *et al.*, 2004; Ban *et al.*, 2011). A number of global studies have been published that focus on modelling MPA operational costs and determining the key variables that underpin the magnitude of operational costs (Balmford *et al.*, 2004; Cullis-Suzuki and Pauly, 2010; Gravestock *et al.*, 2008). Understanding the costs associated with managing MPAs is complicated by the fact that there are often budget short-falls, creating a situation in which the amount spent on MPA management is less than the amount required for effective management (Balmford *et al.*, 2004; Ban *et al.*, 2011). Based on a survey of literature (Balmford *et al.*, 2004; Ban *et al.*, 2011; Blom, 2004; Bruner *et al.*, 2004; Hockings and Phillips, 1999; Hunt, 2013; James *et al.*, 1999; Kuperan *et al.*, 2008; Leisher *et al.*, 2012; Miller *et al.*, 2013; Reid-Grant and Bhat, 2009; Tongson and Dygico, 2004), operational costs may be further broken down into seven different types of cost: administration and management; monitoring, compliance and enforcement; communication; on-going research costs; periodic review; periodic revisions; and off-reserve management (Baulcomb, 2013). For the assessment of the operational costs of expanded MPA coverage we make use of Model 1 from Balmford *et al.* (2004), which relates the operating cost per km² to the area of the MPA (see Table 5). It is noted that

this cost function is based on relatively old data and that new technological developments, particularly regarding the monitoring of activities in MPAs, could bring down the operational costs over time.¹⁴

Data on MPA areas under each scenario are fed into this cost function to estimate the operating costs for each MPA. We assume that these costs are incurred in each year over the period 2020-2050. The estimated costs are converted from 2000 values to 2013 values using a GDP deflator conversion factor from the World Bank World Development Indicators (2015). Table 6 presents the total MPA operating costs for each scenario. Again there are substantial economies of scale, even to the point that the total operating costs are lower under the P2P30 than P2P10 scenario. This occurs due to the agglomeration of smaller (and relatively more costly) MPAs into a smaller number of larger MPAs.

Table 5 MPA operating cost function. Source: Table 1, Balmford et al. (2004)

Variable	Definition	Coefficient
Operating cost	2000 USD/km ² /year; log ₁₀	
Intercept		5.02
MPA area	km ² ; log ₁₀	-0.80

Table 6 MPA operating costs (USD; billions; 2013 price level; present values using discount rate of 3%)

Scenario	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Cost (billions)	41.711	43.080	39.411	42.505	40.798	39.766

In addition to the costs incurred by governments, MPAs may generate costs that are incurred by maritime industries and coastal communities. These costs may be divided broadly into those incurred during the establishment phase and those opportunity costs associated with restrictions on activities during the active management of the MPA. Typically the opportunity costs are larger than the participation costs, though this may not always be the case.

Although there are a number of different sectors that may be impacted by the establishment and operation of MPAs, the opportunity costs most frequently considered are those experienced by the fisheries industry (Hoagland *et al.*, 1995). The opportunity costs incurred by other industries are highly spatially variable, even on fairly small scales (Naidoo and Ricketts, 2006). The research available on this issue is still fairly limited and although our review found individual case studies that discuss the costs to industry of MPA creation (e.g. Cook and Heinen, 2005; Hunt, 2013; Richardson *et al.*, 2006; Adams *et al.*, 2011; Dalton, 2004; Gaines *et al.*, 2010; Jiang *et al.*, 2008; Leathwick *et al.*, 2008; Mangi *et al.*, 2011; Rassweiler *et al.*, 2012; Scholz *et al.*, 2011; Soares-Filho *et al.*, 2010; Stevenson *et al.*, 2013), there are currently no studies that provide statistically verifiable relationships between any facet of MPA design and sector-based opportunity costs (at a local, regional, or global scale).

¹⁴ The Virtual Watch Room developed by The Pew Charitable Trusts and Satellite Applications Catapult provides an example of how satellite tracking and imagery data can be used to reduce monitoring costs of MPAs (Pew, 2015)

Opportunity costs of expanding MPAs on fisheries

Fisheries statistics published by the FAO do not include the value of production by country. Values are provided with respect to trade flows (imports and exports) together with the volumes of those flows. Fisheries export volumes and values (aggregated across commodity types) could be used to estimate the value per tonne of fisheries production on at a country level. However, this is not an ideal measure as it may overestimate the value of fisheries production for a number of reasons: countries may tend to export more commercially valuable species; exported commodities may have undergone processing such as curing or smoking that add value to the landed catch; export data includes re-exported commodities, again these may have value added due to processing. Conversely, it may be the case that the exported commodities are comprised of lower value species including those processed into fish meal and oils. A detailed analysis of trade flows and values is beyond the scope of this analysis.

Instead, FAO estimates of the global value and quantity of capture fisheries were used as an aggregate measure to assess the global impacts of MPA designation. The total value of fisheries production (capture and aquaculture, both marine and freshwater) was estimated to be \$217.5bn in 2010, of which \$125bn was aquaculture, implying that the value of capture fisheries were worth \$98.5bn. The total volume of capture fisheries was 90.4 million tonnes, of which 78.9 million tonnes were from marine fisheries (i.e. 87.3%). If we assume that freshwater and marine fisheries are of similar per unit value, then the value of marine capture fisheries would be approximately \$85.97bn.

There is a lack of equivocal evidence regarding impacts of MPAs on spillovers (i.e. fish stocks and potential harvest increase both within and outwith MPAs); the nature of MPA management and regulation (the degree of no-take or multi-use) and information on how MPA may impact on different species. Although FAO data on fisheries production and value are available at species level these cannot be readily reconciled with the proposed MPA scenarios (hence our aggregated approach). Despite the issues with applying more realistic assumptions with respect to the impact of MPAs on fisheries there are a number of reasons why exploring these is attractive.

An alternative assumption of a linear and proportional response of fisheries to MPA designation implies that existing fishing levels are sustainable. However, if harvest levels are not reduced then many important fisheries could collapse, i.e. fisheries would decline in any case and the potential for spillovers from MPAs could in fact sustain the residual fisheries outside MPAs. Further it is not implicit in the MPA expansion scenarios that these would all be no-take MPAs. Instead there may be a mixture of management approaches impacting on different species in different areas, i.e. a mix of take and no-take with varying levels of restriction. This may be particularly relevant for larger MPAs or those in the Durban (30%) target relative to the Aichi (10%) target.

Figure 8 provides a conceptual representation of potential future pathways for fisheries production under the status quo, 10% and 30% MPA scenarios. Without further MPA designation global fisheries production may see continuing decline in production due to overfishing and stock declines. Under the 10% MPA scenario, there is a consequent decline in production. However, due to spillover impacts and reductions in overall fishing effort the residual rate of harvesting is more sustainable. Although production continues to decline it does so at a decreased rate with the consequence that production eventually exceeds that which would occur without MPA expansion. Under the 30% MPA scenarios there are potentially larger positive spillover

effects outside the MPAs, increasing the possibility of more sustainable fisheries. Consequently, overall production might eventually exceed both the status quo and 10% scenarios.

Extrapolating future trends in fisheries production from existing data is problematic because of different development pathways. The FAO global capture production 1950-2012 data indicate that global production increased steadily from 1950 (17.3 mt) until the 1990s (peaking at 87.7 mt in 1996), an increase of 407%; since then it has declined by 7.8% to reach 80.8 mt in 2012. Of course, such aggregated data does not account for underlying changes in the species being caught or at which trophic levels; arguably as lower trophic levels are targeted, the overall sustainability of fisheries is put at risk as marine food webs become less resilient.

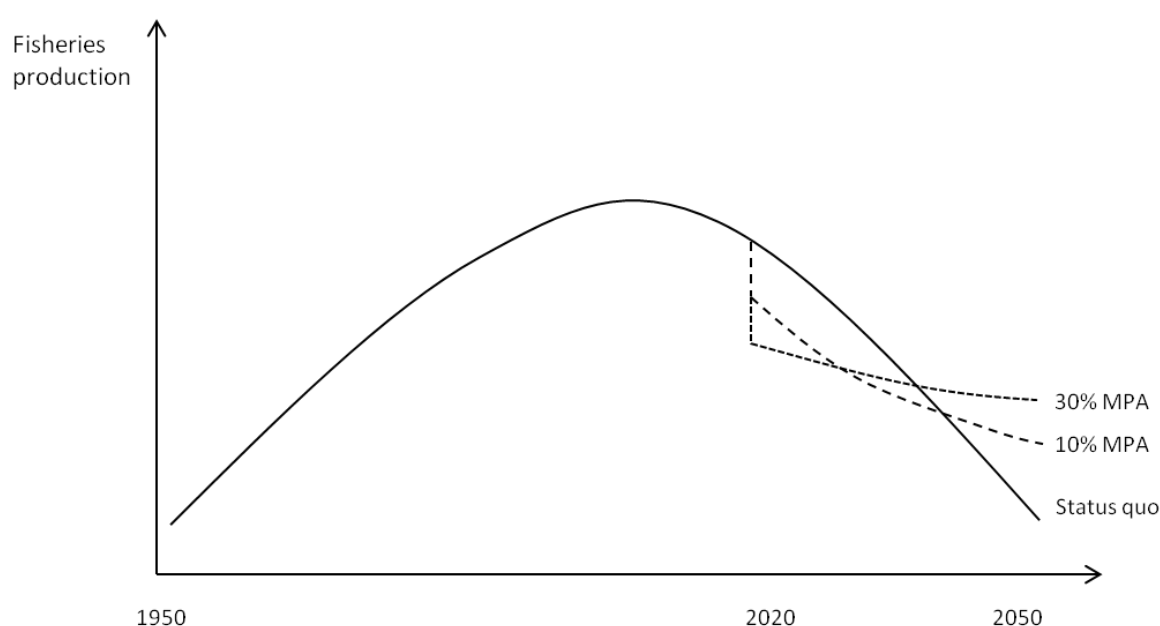


Figure 8 *Conceptual representation of the potential impact of MPA expansion on fisheries production*

To test the sensitivity of the fisheries opportunity costs to assumptions about the impact of MPA designation we examine the following three options:

1. No MPA expansion (status quo): fisheries values decline by 1%, 2% or 4% per annum from 2015 to 2050.
2. 10% MPA expansion (for each of the relevant scenarios): fisheries values decline at 1%, 2% or 4% per annum until 2020 (MPAs established), the area of the new MPAs are no-take and spillovers reduce the decline in fisheries outside the MPAs to 0.5%, 1% or 2% per annum.
3. 30% MPA expansion (for each of the relevant scenarios): fisheries values decline at 1%, 2% or 4% per annum until 2020 (MPAs established), the area of the new MPAs are no-take together with spillovers this reduces decline in fisheries outside the MPAs to 0.2%, 0.5% or 1% per annum.

The chosen percentage reductions in fisheries values are arbitrary as the global impact of MPA networks on fisheries production across a variety of species at different trophic levels is highly uncertain and would depend on specific management measures.

Furthermore, as we are dealing with values, even if impacts on fisheries in volume terms were known there is further uncertainty with respect to prices and consequently overall value. Large reductions in quantity produced would be expected to be associated with increases in price; although of course collapse of individual fisheries would reduce values to zero (thresholds are a further source of uncertainty). Table 7 presents the estimated outcomes of the trajectories outlined above for each of our exploratory scenarios for MPA expansion.

Table 7 illustrates how sensitive net costs to fisheries are to both the assumptions made about the decline in fisheries in the absence of MPAs and the extent to which MPAs can mitigate that decline. If a low decline in fisheries of 1% per annum without additional MPAs is assumed then the designation of additional MPAs would result in net losses (opportunity costs) based on the assumptions made about the reductions in fisheries decline for the MPA scenarios. However, as the potential decline in fisheries increases, the potential positive impacts of MPAs also increase. Essentially, the higher the potential decline in fisheries due to over-exploitation then the greater the benefits from introducing MPAs as these reduce the decline in fisheries outside MPAs through spillovers. Although this analysis has been applied to global fisheries production it indicates that MPA designation and management should be targeted where potential declines in fisheries are highest.

The results in Table 7 indicate that within each of the fisheries decline trajectories there is little variation in impact across the 10% MPA scenarios. For the 30% MPA coverage scenarios E2E30 and P2P30 are also similar and result in considerably lower benefits (or higher losses) than the P2M30 scenario. The higher net benefits (or lower losses) associated with the 10% MPA scenarios indicate that the value of global fisheries remain highest under lower expansion of MPAs. This suggests that (under our assumptions) additional positive spillovers from expanding to 30% MPA coverage would not compensate for additional fishing restrictions for any given fisheries decline trajectory. However, in the case that global fisheries face a high rate of decline (4% annually under the status quo), both the 10% and 30% scenarios for MPA coverage lead to net fisheries benefits as compared to taking no additional marine protection.

In summary, the expansion of MPA coverage results in net costs to fisheries if the baseline rate of decline in fisheries production is low; but could result in net benefits, through positive spillover effects, if the baseline rate of decline in production is high.

Table 7 Value of global marine fisheries under MPA scenarios for different trajectories of reduction in fisheries values with sensitivity to different rates of fisheries decline (USD; billions; 2013 price level; present values using a discount rate of 3%)

	Annual reduction in fisheries production	MPA expansion scenarios					
		E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Low decline							
Status quo	1%	1690.6	1690.6	1690.6	1690.6	1690.6	1690.6
10% MPA	0.5%	1670.7		1669.0		1670.5	
30% MPA	0.2%		1493.2		1525.3		1494.8
Net benefit		-19.8	-197.3	-21.5	-165.2	-20.0	-195.8
Medium decline							
Status quo	2%	1493.7	1493.7	1493.7	1493.7	1493.7	1493.7
10% MPA	1%	1561.4		1559.8		1561.2	
30% MPA	0.5%		1430.3		1460.8		1431.8
Net benefit		67.7	-63.3	66.2	-32.8	67.6	-61.8
High decline							
Status quo	4%	1199.1	1199.1	1199.1	1199.1	1199.1	1199.1
10% MPA	2%	1373.4		1372.1		1373.2	
30% MPA	1%		1330.7		1358.7		1332.1
Net benefit		174.3	131.7	173.0	159.6	174.2	133.0

4.3 Benefits

The economic benefits of establishing and operating MPAs are the maintained or enhanced flows of ecosystem services that are provided by protected marine ecosystems (Sala *et al.*, 2013; Potts *et al.*, 2014). Key marine ecosystem services include: the provision of food for subsistence or commercial use; tourism and recreation; coastal protection; carbon sequestration; and biodiversity.

By providing appropriate food and habitat conditions, mangrove, seagrass, reef, and open sea ecosystems support the growth and reproduction of a range of fish and invertebrate species that can be used as food for humans. Subsistence food refers to the extraction of fish, invertebrates, and other food goods for consumption within the households of those harvesting the food goods. Commercial food refers household-scale and industrial-scale harvesting of fish and other food goods for sale locally, regionally or internationally.

Tourists flock to coastal destinations for a number of reasons, including their aesthetic beauty and opportunities for marine-based activities. Tourism can also have negative impacts on the marine environment resulting from pollution, disturbance and direct damage to by visitors to marine ecosystems. The case study on the Galapagos Islands protected areas (section 5.5) provides a comparison of the net benefits of alternative tourism development paths. Marine and coastal ecosystems also provide opportunities for recreation by local residents. Local recreational activities may be less visible in markets but nevertheless contribute substantially to human well-being.

Coastal areas can be vulnerable to flooding and erosion from tidal currents and wave action. Mangroves, coral reefs and seagrass beds provide protection from damaging waves and storm surges. The protection of human lives and assets is an ecosystem service (van Zanten *et al.*, 2014).

Mangroves, wetlands, coral reefs, seagrasses, phytoplankton and algae all remove carbon dioxide from the atmosphere and store it in their fibres, in the soil, and in the ocean substrate. The amount of carbon that is captured from the atmosphere by different plant species can be quantified in terms of a rate of sequestration. If a tree or plant is destroyed, the carbon stored in the plant's cells is released as the biomass decays or burns. Carbon stored in the soil/substrate may be released over time if left un-vegetated, or released quickly if the substrate is disturbed. Both the rate at which carbon is added to biomass/substrate and any release of stored carbon are important. Together they represent the net carbon dioxide sequestered from the atmosphere, or change in the stock of stored carbon (Pendleton, 2012).

Marine and coastal ecosystems house remarkable biological diversity that may play multiple roles in the provision of ecosystem services (Mace *et al.*, 2012); firstly as a supporting or intermediate service in maintaining ecosystem resilience underlying the provision of many final services, and secondly as final ecosystem services in the form of diverse biological compounds with human applications or non-use values that people hold unrelated to any current or future use.

Bio-prospecting is the process of discovering and commercializing new products from natural sources. Marine resources, particularly areas with high biodiversity such as coral reefs or unique ecology such as deep-sea thermal vents, may house potentially marketable products or elements that lead to marketable products.

The appreciation individuals have for ecosystems, even when they are not directly or indirectly using the ecosystem, is also an ecosystem service. Individuals may simply enjoy knowing that an ecosystem exists (existence value), or they may appreciate knowing that a resource will be available for future generations (bequest value) or for future uses that have not yet been realized (option value) (Borger *et al.*, 2014; Jobstvogt *et al.*, 2014). See section 5.1 for a case study on the valuation of ecosystem services for Bonaire, which includes an assessment of non-use values.

The contribution of ecosystems in building social capital is also recognised as a cultural ecosystem service (Chan *et al.*, 2012). Social capital is broadly defined as the social relationships and cohesion between individuals and communities that encourage reciprocity and exchanges, and enable the establishment of common rules, norms and sanctions. Ecosystems may play a role in building social capital by providing space and opportunities for social interaction. Ecosystems may also play a role in establishing and maintaining cultural identity (Barnes-Mauthe *et al.*, 2015).

Marine and coastal ecosystems support a number of other important ecosystem services, including provision of raw materials (e.g. sand), pollution remediation, oxygen generation, temperature regulation, primary production, and other supporting ecosystem services.

The beneficiaries of marine ecosystem services are diverse and range from local coastal communities to future global populations (in the case of carbon sequestration and climate change mitigation). See sections 5.2, 5.4 and 5.6 for case studies highlighting the benefits of marine ecosystem services to local communities (in the case of Fiji locally managed marine areas), distant populations (in the case of the Sargasso Sea) and multiple beneficiaries (Arctic case study).

Coral reef ecosystem services

For the assessment of the benefits of expanding MPA coverage we make use of published value functions for marine ecosystem services. A value function for coral reef ecosystem services is obtained from Hussain *et al.* (2011) and is presented in Table 8. This value function is estimated from a sample of 163 value estimates taken from 72 primary studies that cover a broad set of ecosystem services, including recreational diving, recreational snorkelling, recreational fishing, other tourism activities, commercial fisheries, coastal protection, research and non-use values for biodiversity. The value function is used to predict the value per unit area (hectare) for a bundle of ecosystem services, reflecting the extent of provision of each service observed in the underlying primary valuation studies.

The explanatory variables included in the coral reef value function are as follows: the area of coral cover; the GDP per capita of the country in which the reef is located; the population within a 50km radius; the length of roads within a 50km radius; the human appropriation of net primary product within a 50km radius; the net primary product within a 50km radius; and the area of coral cover within a 50km radius. A GIS is used to construct a global database of coral reefs that contains data on each of these variables. The scenario maps described in section 4.1 are then overlain to identify which coral reefs are covered by MPAs in each scenario.

Table 8 Coral reef value function. Source: Table 32, Hussain *et al.* (2011)

Variable	Units*	Coefficient
Value of ecosystem services (dependent)	USD/ha/year; 2007; ln	
Intercept		16.093
Area of coral cover	ha; ln	-0.293
GDP per capita	2007 USD; ln	0.039
Population within 50km	population; ln	0.238
Area of coral reef within 50km	ha; ln	-0.207
Length of roads within 50km	km; ln	-0.035
Net primary production within 50km	tonnes; ln	-0.379
Human appropriation of net primary production within 50km	tonnes; ln	-0.076

* ln denotes the natural logarithm.

Baseline rates of loss of coral cover are assumed to continue at an average of 2% per year (Bruno and Selig, 2007) but are distributed around this value to reflect spatial variation in risk derived from the Reefs at Risk Revisited study (Burke *et al.*, 2011). We assume that a low risk rating translates to an annual rate of loss of 1%; medium to 2%; high to 2.5% and very high to 3%. These annual rates are applied to the global database of coral reefs to compute future areas of coral cover under the baseline. The impact of MPA expansion under each scenario is assumed to be a 20% increase in coral cover relative to the baseline (Magdaong *et al.* 2014), if a reef is located within an MPA. Differences in coral cover between the baseline and each scenario are computed. This assessment assumes that rates of loss continue on a linear path. It may be the case, however, that thresholds and tipping points in coral ecosystems result in higher and non-linear rates of loss.

Baseline changes in the values of explanatory variables in the value function are also computed to 2050. The area of other coral reefs within 50km of each target reef is computed in a similar manner to changes in reef area. GDP and population growth rates are obtained from the OECD (2012; 2014). Rates of road infrastructure

development are obtained from the IMAGE-GLOBIO model (Alkemade *et al.*, 2009; PBL, 2010).¹⁵

Data for each coral reef are fed into the value function to estimate the marginal value of changes (improvements) in coral cover relative to the baseline. The aggregated benefits of improved provision of coral reef ecosystem services for each scenario are presented in Table 9. The estimated benefits of MPA protection are substantial, reflecting both the high economic value of coral reef ecosystem services and the high rates of loss of coral cover in the absence of conservation intervention. The results also show very large differences in the yield of benefits across scenarios. The spatial distribution of MPAs under the P2P scenario (i.e. targeting areas with high biodiversity and low human impact) delivers much higher returns.

Table 9 Benefits of improvement in the provision of coral reef ecosystem services (USD; billions; 2013 price level; present values using discount rate of 3%)

Scenario	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Coral benefits	108.848	164.135	214.218	428.451	410.443	521.198

Coastal wetland ecosystem services

A value function for coastal wetland ecosystem services is obtained from Hussain *et al.* (2011) and is presented in Table 10. This value function is estimated from a sample of 247 separate value estimates from 131 primary studies that cover a broad set of ecosystem services, including flood protection, water supply, water quality, habitat and nursery for fauna, recreational hunting, recreational fishing, food and material provisioning, fuel wood provisioning, non-consumptive recreation, aesthetic enjoyment and biodiversity conservation. The value function is used to predict the value per unit area (hectare) for a bundle of ecosystem services, reflecting the extent of provision of each service observed in the underlying primary valuation studies.

The explanatory variables included in the wetland value function are as follows: the area of wetland; the GDP per capita of the country in which the wetland is located; the population within a 50km radius; the area of lakes and rivers within a 50km radius; the area of other wetlands within a 50km radius; the population within a 50km radius; and the human appropriation of net primary product within a 50km radius. A GIS is used to construct a global database of coastal wetlands that contains data on each of these variables. The scenario maps described in section 4.1 are then overlain to identify which coastal wetlands are covered by MPAs in each scenario.

¹⁵ GLOBIO is a modelling framework developed to calculate the impact of five environmental drivers on biodiversity. GLOBIO is based on cause-effect relationships derived from the literature and uses spatial information on environmental drivers as input. This input is mainly derived from the Integrated Model to Assess the Global Environment (IMAGE). Projections for environmental drivers are based on the OECD Environmental Outlook (OECD, 2008) and cover the period 2000–2050.

Table 10 Wetland value function. Source: Table 29, Hussain *et al.* (2011).

Variable	Units	Coefficient
Value of ecosystem services	USD/ha/year; 2007; ln	
Intercept		1.708
Area of wetland	ha; ln	-0.209
GDP per capita (PPP US\$ 2007)	2007 USD; ln	0.610
Area of lakes and rivers within 50km	ha; ln	0.159
Area of wetlands within 50km radius	ha; ln	-0.175
Population within 50km	population; ln	0.426
Human appropriation of NPP within 50km	tonnes; ln	-0.201

Baseline rates of loss of coastal wetland are assumed to continue at 1.5% per year (Pendleton *et al.*, 2012). This annual rate is applied to the wetland data to compute future areas of wetland under the baseline. Wetland loss is assumed to fall to zero if a wetland is located within an MPA. It may be the case that wetland area increases under protection but we are not able to assess this. Differences in wetland area between the baseline and each scenario are computed. Baseline changes in the values of explanatory variables in the value function are also computed to 2050 using the same data and source as for the coral analysis.

Data for each wetland site are fed into the value function to estimate the marginal value of changes (increases) in wetland area relative to the baseline. The aggregated benefits of improved provision of wetland ecosystem services for each scenario are presented in Table 11.

Table 11 Benefits of improvement in the provision of coastal wetland ecosystem services (USD; billions; 2013 price level; present values using discount rate of 3%)

Scenario	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Wetland benefits	415.067	450.212	367.871	455.347	407.330	515.447

Mangrove ecosystem services

For mangroves, separate methods are used for estimating the value of carbon storage and other ecosystem services. The reason for treating carbon storage separately is that as a global pollutant, the unit value or price of carbon emissions does not vary spatially, whereas the unit values of other ecosystem services are highly spatially variable and require the use of value function that reflects this. The value function for mangrove ecosystem services is obtained from Brander *et al.* (2012a) and is presented in Table 12. This value function is estimated from a sample of 130 value estimates taken from 48 primary studies that cover a broad set of ecosystem services, including coastal protection, fisheries, fuel wood provisioning and water quality regulation. The value function is used to predict the value per unit area (hectare) for a bundle of ecosystem services, reflecting the extent of provision of each service observed in the underlying primary valuation studies.

The explanatory variables included in the mangrove value function are: dummy variables indicating the presence or absence of four ecosystem services (coastal protection, water quality improvement, support for fisheries, and fuel wood); the area of the mangrove patch; total area of mangrove within 50km radius; the length of roads

within a 50km radius; the GDP per capita of the country in which the mangrove is located; and the population within a 50km radius. A GIS is used to construct a global database of mangroves that contains data on each of these variables. The scenario maps described in section 4.1 are then overlain to identify which mangroves are covered by MPAs in each scenario.

Table 12 Mangrove value function. Source: Table 2, Brander et al. (2012a)

Variable	Variable definition*	Coefficient
Value of ecosystem services (dependent)	USD/ha/year; 2007; ln	
Intercept		-0.590
Dummy variable for coastal protection ES		1.456
Dummy variable for water quality ES		1.714
Dummy variable for fisheries ES		0.860
Dummy variable for fuel wood ES		-1.085
Area of mangrove	ha; ln	-0.343
Total area of mangroves within 50 km	km ² ; ln	0.248
Length of roads within 50 km	km; ln	-0.312
GDP per capita (USD; ln)	2007 USD; ln	0.785
Population within 50 km	population; ln	0.284

* ln denotes the natural logarithm

Baseline rates of loss of mangrove are assumed to fall within the range 0.7-3% per year (Pendleton *et al.*, 2012) and are distributed within this range reflecting the spatial variation in risk derived from the Reefs at Risk Revisited study (Burke *et al.*, 2011). The reasoning behind this assumed correspondence between risk to reefs and risk to mangroves is that the population and development pressures that drive differences in risk to coral will also tend to drive degradation of mangroves. We recognise, however, that there are differences in the way in which development pressures affect different biomes, and indeed that mangroves face unique pressures (Duke *et al.*, 2007). We assume that a low risk rating translates to an annual rate of loss of 0.7%; medium to 1.9%; high to 2.5% and very high to 3%. These annual rates are applied to the mangrove data to compute future areas of mangrove under the baseline. Mangrove loss is assumed to fall to zero if a mangrove is located within an MPA. It may be the case that mangrove area increases under protection but we are not able to assess this. Differences in mangrove area between the baseline and each scenario are computed. Baseline changes in the values of explanatory variables in the value function are also computed to 2050 using the same data and source as for the coral analysis.

Data for each mangrove site are fed into the value function to estimate the marginal value of changes (increases) in mangrove area relative to the baseline. The aggregated benefits of improved provision of mangrove ecosystem services for each scenario are presented in Table 13.

The value of avoided carbon emissions and additional sequestration by mangroves is estimated using the methods and parameters described in Pendleton *et al.* (2012) and Murray *et al.* (2011). The quantities of avoided carbon emissions from mangrove biomass and soil are computed using estimates of stored carbon, tCO₂/ha 563 and tCO₂/ha 1,800 respectively. Following Murray *et al.* (2011) we assume that 75% of biomass carbon is released immediately and that the remaining 25% decays with a half-life of 15 years; and that soil organic carbon has a half-life of 7.5 years. We combine these parameter values with the avoided loss in mangrove area in each year of the

analysis to compute the quantity of avoided emissions in each year. The quantities of additional carbon sequestration due to the avoided loss of mangroves is estimated using a sequestration rate of $\text{tCO}_2/\text{ha}/\text{year}$ 6.3. We multiply this rate by the cumulative area of avoided mangrove loss in each year of the analysis. The estimated quantities of avoided carbon emissions and additional sequestration in each year are then multiplied by year specific estimates of the social cost of carbon (Interagency Working Group, 2013). These estimates are converted to 2013 price level using a GDP deflator derived from the World Development Indicators (World Bank, 2015). The present value of mangrove carbon benefits under each scenario is presented in Table 13.

The estimated benefits of MPA protection in terms of mangrove ecosystem services are again substantial. The value of avoided carbon emissions and increased carbon sequestration represents a substantial proportion of the total benefits obtained by protecting mangroves (approximately 40%). It is notable that there is relatively little variation in estimated benefits across scenarios; also that expanding the extent of MPA coverage from 10% to 30% of marine area does not yield much increase in benefits from mangrove ecosystem services.

Table 13 Benefits of improvement in the provision of mangrove ecosystem services (USD; billions; 2013 price level; present values using discount rate of 3%)

Scenario	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Non-carbon services	70.055	75.062	67.726	72.078	75.027	77.902
Carbon storage	27.975	29.809	27.156	28.913	29.738	30.622
Total	98.030	104.871	94.882	100.991	104.766	108.524

4.4 Cost-Benefit Analysis

The results of the cost-benefit analysis are presented in Table 14. Under all scenarios, the expansion of MPAs has a positive benefit-cost ratio (in the range 3.17 – 19.77). In the case of the P2P10 scenario (targeting areas with high biodiversity and low human impact with up to 10% coverage of total marine area), each dollar invested yields a return of around 20 dollars in benefits. The corresponding internal rates of return for each scenario are all greater than the discount rate used in the analysis (in the range 9–24%). The net improvement in human welfare, as measured by the net present value (NPV) of each scenario, is estimated to be in the range USD 490–920 billion over the period 2015–2050. On this evidence, investing in MPAs is economically advisable.

The results also show that there are substantial differences between the scenarios, indicating that the scale of expansion and targeted locations of MPAs makes a considerable difference to their economic performance. The E2E10 scenario (targeting low biodiversity and low human impact areas with up to 10% coverage of total marine area) has the lowest costs (and in that sense lives up to its title “easy-to-expand”) but also yields the lowest benefits. Creating MPAs to simply meet the spatial requirements of the Aichi and Durban Targets at lowest cost will result in positive net returns but will also mean missing the opportunity to obtain much higher benefits from marine ecosystem services. Pursuing an expansion of MPA coverage that targets areas of high biodiversity yields substantially higher returns.

The results also reveal the presence of diminishing returns to scale from expanding MPAs. Under all scenarios, expanding MPAs from 10% to 30% coverage of total marine area yields higher absolute net benefits but at a declining rate. This is reflected by the lower benefit-cost ratios under 30% coverage, as compared to the corresponding 10%

coverage scenarios. The factors underlying diminishing returns to scale in this analysis are that high value marine habitats are already protected under the 10% cover scenarios and that the opportunity costs to fisheries increase more than proportionately with increasing MPA cover. The establishment and operational costs of MPAs decline with scale but these constitute a smaller share of total costs.

It is important to note that this analysis contains only a partial view of the full set of costs and benefits associated with expanding MPAs. On the costs side we are missing information on the opportunity costs of other marine activities such as mineral extraction and energy generation. On the benefit side we are missing information on the impacts to all ecosystems (e.g. seamounts, seagrass, kelp forests) and all ecosystem services (e.g. bio-prospecting and existence values associated with marine biodiversity) that are potentially positively affected by MPAs. On balance, we expect that adding further information would tend to increase the benefits of expansion relative to costs since existing estimates for non-use values for marine biodiversity are generally high (Borger *et al.*, 2014; Jobstvogt *et al.*, 2014).

It should also be noted that the globally aggregated results presented in Table 15 provide an indication of the economic performance of each scenario as a whole. At the national level, and to a greater extent at the level of individual MPAs, there is likely to be much wider variation in returns from expanding MPAs.

Table 14 Cost-Benefit Analysis of expanding MPAs (USD; billions; 2013 price level; present values using discount rate of 3%)

	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Man. costs	19,465	23,768	18,563	38,039	20,824	24,374
Est. costs	5,330	5,980	5,022	20,188	5,842	7,854
Fisheries	19.8	197.3	21.5	165.2	20	195.8
Total costs	44.595	227.048	45.086	223.426	46.666	228.027
Coral ES	108,848	164,135	214,218	431,717	410,443	521,198
Mangrove ES	70,055	75,062	67,726	72,078	75,027	77,902
Mangrove carbon	27,975	29,809	27,156	28,913	29,738	30,622
Wetland ES	415,067	450,212	367,871	455,347	407,330	515,447
Total benefits	621,945	719,218	676,970	988,055	922,538	1145,169
NPV	577,351	492,169	631,885	764,628	875,872	917,141
BCR	13.95	3.17	15.02	4.42	19.77	5.02
IRR	21%	9%	21%	11%	24%	11%

4.5 Employment

Employment generated by a policy or investment is often an important point of consideration for decision makers. From an economic perspective employment represents a cost of policy implementation (i.e. the labour employed cannot be productively used elsewhere in the economy). From a political and societal point of view, however, the number of jobs created by a policy may be viewed as a positive aspect, particularly in regions or sectors with persistent unemployment. In order to provide information on this aspect of expanding MPAs, we use the employment

function estimated in Balmford *et al.* (2004) to estimate the number of full-time jobs in MPA management generated under each scenario. The employment function is presented in Table 15 and the estimated number of jobs is presented in Table 16. The numbers of jobs created also show diminishing returns to scale, with relatively few additional jobs created through expanding MPA coverage from 10% to 30%. In the case of the P2P scenario, the total number of jobs created even decreases with expansion from 10% to 30% because multiple small MPAs become consolidated into fewer larger MPAs and require fewer staff. These are full-time jobs that are directly related to MPA management. In addition, a wide variety of jobs may be created indirectly in related sectors (e.g. tourism). See section 6.3 for a case study on the direct and indirect employment related to the Great Barrier Reef Marine Park. It may also be the case that the expansion of MPAs leads to losses in other maritime industries such as fisheries and oil and gas extraction.

Table 15 MPA employment function. Source: Table 1, Balmford *et al.* (2004)

Variable	Definition	Coefficient
Full time jobs	jobs/km ² ; log ₁₀	
Intercept		0.85
MPA area	km ² ; log ₁₀	-0.77

Table 16 Number of full-time jobs in MPA management

Scenario	E2E10	E2E30	P2M10	P2M30	P2P10	P2P30
Employment	165,138	167,695	156,542	183,474	162,750	156,281

5 Case studies

5.1 The hidden values of the Bonaire Marine Park

Bonaire is an island in the Caribbean located north west of Venezuela and is formally a special municipality of the Netherlands. Bonaire spreads across 288 sq-km and is famous for its coral reefs. The entire coastline of Bonaire is protected, granted with the status of Marine Park established in 1979. The value of Bonaire's ecosystems to tourism has been assessed to be much higher than in other comparable sites. Bonaire is very popular among the diving community and is consistently ranked in the Top 3 of the Scuba Diving Magazine. As a result, the economy of Bonaire relies greatly on dive tourism.

Another distinctive feature of the Bonaire Marine Park is the high non-use value of the marine ecosystem. An extensive survey among citizens of the Dutch mainland revealed a high appreciation of Bonaire's ecosystems, even if these people have no intention of visiting the island. The average monthly amount that residents of the Netherlands are willing to pay for nature protection in the Caribbean Netherlands is estimated at around USD 7 per household, which aggregates to a total of USD 60 million. As shown in Figure 9, this constitutes the largest component of the Total Economic Value of Bonaire's ecosystems.

This evidence of the willingness to pay of Dutch mainland citizens for nature conservation in the Caribbean Netherlands built an argument for securing a €7.5 million investment for nature conservation on the three Dutch islands by the Ministry of Economic Affairs. Also WWF Netherlands used the study results to allocate a budget for conservation efforts in the Caribbean Netherlands.

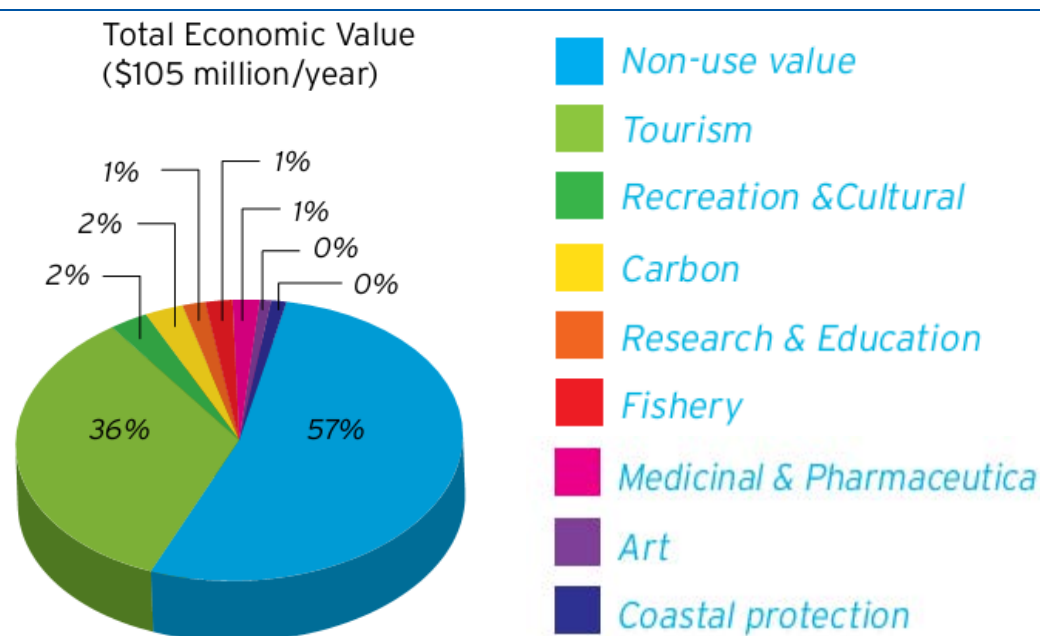


Figure 9 The contribution of non-use values to the Total Economic Value of the ecosystems of Bonaire

5.2 Fiji Locally-Managed Marine Area Network as Natures Investment Bank

Alarmed by the drastic decline in marine resources, a community on the eastern coast of Fiji's largest island established the first locally managed marine area (LMMA) in Fiji in 1997. Seven years later, the clam populations had rebounded and household incomes had risen notably with increased harvests. The success of the LMMA in this single village spread rapidly. By 2009, the network had increased to include some 250 LMMAs, covering more than 25% of Fiji's inshore area, and also inspired replication in countries other Pacific countries (United Nations Development Programme, 2012).

A study on the role of MPAs in alleviating poverty proved that the popularity of the concept of LMMAs is entirely based on the fact that this type of conservation translates into a multitude of benefits for local communities (Van Beukering *et al.* 2013). As noted by a Fijian community leader "the marine protected area (MPA) is like a bank to the people: by conserving marine resources, people will reap higher returns in the future". As partly shown in Figure 10, the study provides clear evidence that poverty had been reduced by several factors, going far beyond economic gains: (i) improved fish catches; (ii) new jobs, mostly in tourism; (iii) stronger local governance; and (iv) benefits to health and women.

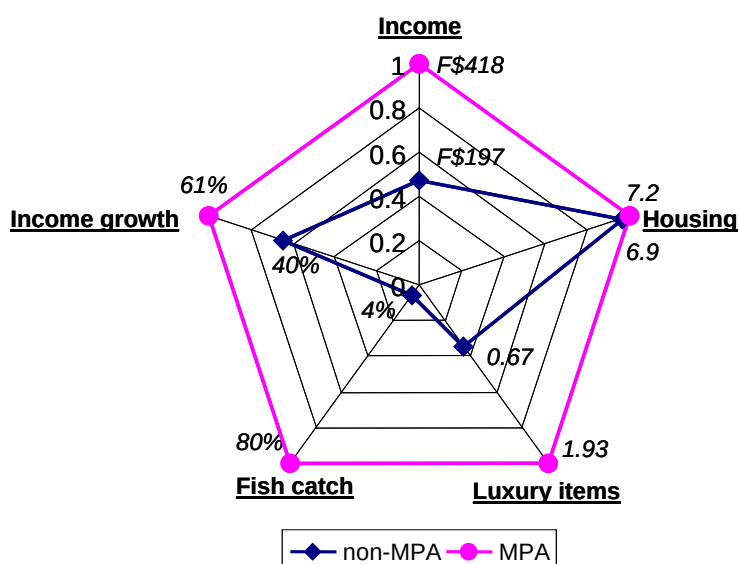


Figure 10 Diamond graph showing how the Fijian villages that established LMMAs score in welfare assets in comparison to villages that did not have an LMMA. Source: Van Beukering *et al.* (2013). p. 124

Another factor contributing to the rapid uptake and replication of the LMMA approach is the relatively low cost of creating and managing a site. For instance, the total cost of establishing Navakavu LMMA was estimated at less than USD 12,000 over five years, a modest investment that has led to a doubling of average household income for about 600 people. A separate study in Navakavu showed that the increase in fish caught over the same timeframe provided about USD 37,800 in benefits to the community. As finfish and invertebrate stocks continue to grow, it is expected that local LMMA benefits will continue to increase (O'Gara 2007).

5.3 Employment gains and losses in the Great Barrier Reef Catchment

Being the largest coral reef ecosystem and one of the seven natural wonders of the world, the Great Barrier Reef is truly famous around the globe. This unique marine ecosystem has also drawn the attention of economists, who have estimated the contribution of the Great Barrier Reef Marine Park to the economy of Australia for the period 2004 and 2013. The latest studies estimate the added value of the Great Barrier Reef at AU\$5.7 billion, which is predominantly based on tourism benefits (Deloitte Access Economics 2013).

What distinguishes these studies from most other economic valuation studies on marine protected areas around the world is the fact that effort has been put into estimating the employment benefits generated through marine ecosystem services. As shown in Table 17 the direct and indirect employment resulting from the Great Barrier Reef services is estimated at 47,615 and 21,364 respectively.

This economic benefit is particularly interesting in the context of recent plans to expand the port at Abbot Point in northern Australia for the export of coal. This plan is mainly promoted under the premise of boosting the economy and the creation of jobs. The plan involves dredging three million cubic meters of sand and mud to be dumped elsewhere, inside the marine park. Various experts claim this could have a disastrous impact on the reef. As a result, the jobs created through the port expansion may well be lost as a result of the decline of the ecosystem services provided by the Great Barrier Reef.

Table 17 Employment generated through ecosystem services of the Great Barrier Reef

Sector	Stay-over tourism	Commercial fishing	Recreation	Research	Total
Direct employment	44,851	533	1,767	464	47,615
Indirect employment	19,487	442	1,018	417	21,364
Total employment	64,338	975	2,785	881	68,979

Source: Deloitte Access Economics 2013

5.4 Short- and long-distance services of the Sargasso Sea

Except for the territory of Bermuda and the Bermudian Exclusive Economic Zone, the Sargasso Sea lies in an area beyond national jurisdictions, known as the high seas. Ocean currents, global biochemical cycles, and wide-ranging ecological processes imply that the influence of the Sargasso Sea is felt within and well beyond its own boundaries.

The ecosystems provided by the Sargasso Sea vary widely in terms of type and beneficiary. Some of its services may be harvested directly (e.g., fish). Other ecosystem elements provided by the Sargasso Sea, such as Sargassum – a floating sea plant, supports part of the life cycle of organisms that ultimately benefit people far from the region. For example, eels that spawn in the Sargasso Sea are later harvested in North America and Europe. The Sargasso Sea also provides important habitat for whales and turtles that return to near shore, continental waters where they support local tourism industries. The Sea also generates non-use and regulating services that benefit people globally.

A recent study provides the best available information about the potential economic magnitude or nature of the Sargasso Sea's ecosystem services (Pendleton *et al.* 2014). The study concludes that economic impacts and benefits directly or potentially linked to the Sargasso Sea may total between tens to hundreds of million dollars a year (see Figure 11). The findings show the ecological health of the Sargasso Sea is not only in the interest of the inhabitants of Bermuda. Better management, including marine protection, of the Sargasso Sea would benefit people and businesses around the globe, in particular, in North America (whale watching), Europe (eel fishing), and elsewhere in the Americas (commercial fishing).

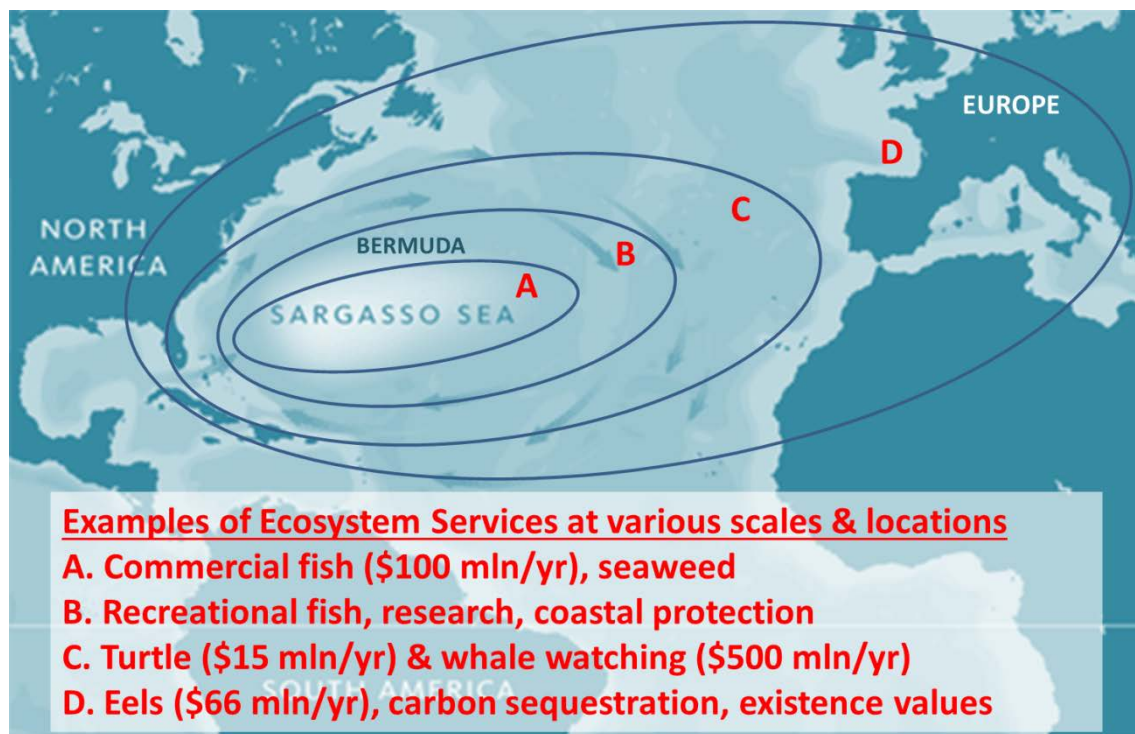


Figure 11 Ecosystem services benefiting people in various locations

5.5 Balancing growth in the Galapagos Islands

The Galapagos Islands have two protected areas: the terrestrial park established in 1959 and the marine reserve established in 1998, which covers approximately 138,000 square kilometres. With only 2000 annual visitors in the 1960s, the tourism industry on the Galapagos Islands has grown by a factor 100 in last 50 years. With more than 200,000 visitors in 2013, the tourism industry is currently the engine of the economy of the Galapagos. This puts significant pressure on the islands' pristine ecosystems, the same ecosystems that attract the tourists to the archipelago. Therefore, balancing the environmental impacts of tourism and the benefits it brings to the islands is a major challenge.

A recent study shows that unlimited growth of tourist numbers leads to lower economic gains in the long term (Schep *et al.*, 2014). For example, the share of visitors that intend to return to the Galapagos reduces from 60% under current conditions to 20% in case of a doubling of crowds, and to less than 10% if the environment degrades substantially. This has implications for the tourism industry and thereby on the prosperity of the archipelago in general.

As part of the study, different tourism growth scenarios were analysed using an extended cost-benefit analysis (CBA) incorporating economic and environmental costs and benefits. The two main scenarios involve a moderate growth from the 200,000 visitors in 2013 to less than 300,000 in 2025, and a rapid growth scenario that results in a 500,000 visitors in 2025. Figure 12 presents the tourism arrivals and corresponding economic net benefits for each scenario. In the short term, the economy can benefit from strong growth in the tourism sector. However, these benefits are not likely to be sustained in the longer run. A stable number of visitors or moderate growth up to 300,000 visitors is more likely to be beneficial for the prosperity of the Galapagos in the long run. The tourism industry on the Galapagos has grown excessively in the last decades. A tourism plan that controls growth within the ecological limits of the Galapagos will provide the highest benefits for the economy.

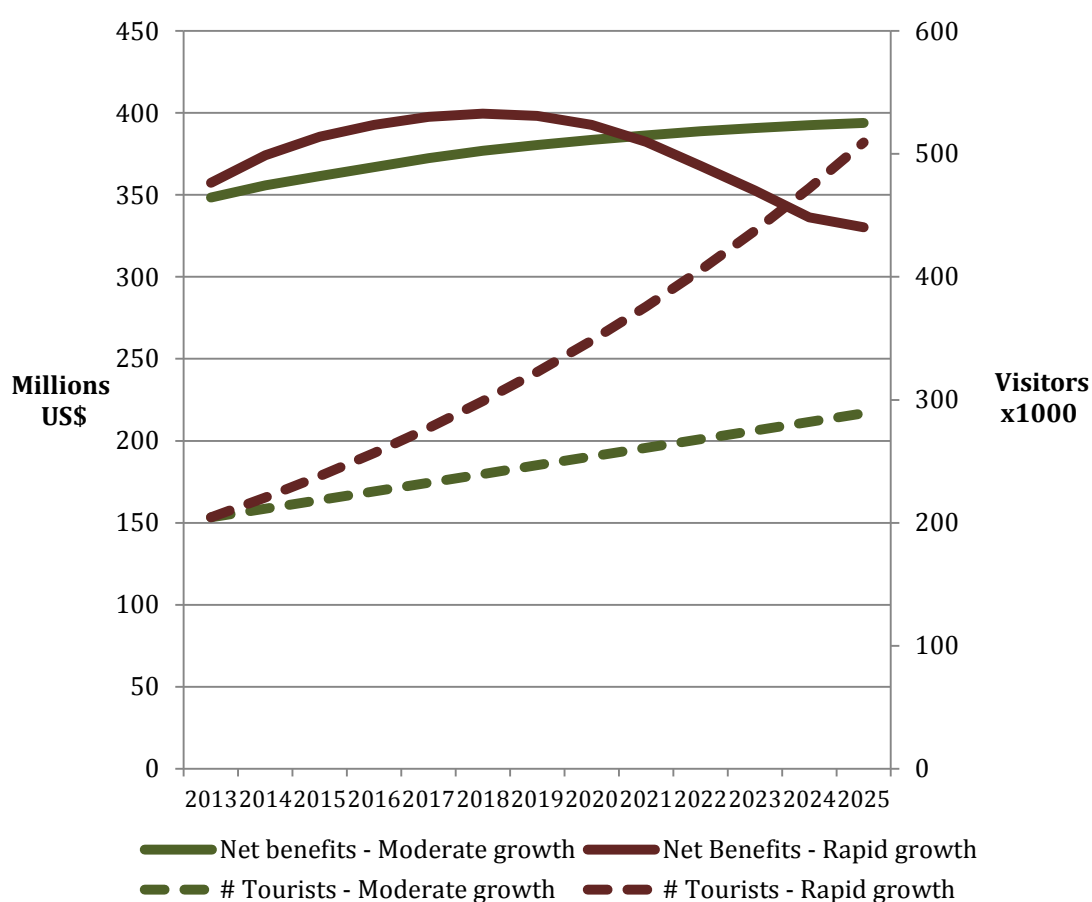


Figure 12 Analysis of two tourism growth scenarios on the Galapagos Islands

5.6 Valuing the invaluable Arctic: work in progress¹⁶

The Arctic is in many people's minds one of the last great wildernesses in the world; cold, hostile, deserted and strangely exotic. To its roughly four million inhabitants, and especially the indigenous populations among those, it is nothing like this and at

¹⁶ Prepared by Miriam Geitz and Pieter van Beukering

the same time so much more. So what the Arctic means to you depends hugely on the eye of the beholder. Among the wide range of arctic ecosystem services, the ones considered of most obvious value are “provisioning services”, the supply of food to people, such as fish, seals or reindeer. These services also provide less obvious benefits to the people harvesting the food, such as cultural and spiritual services connected to harvesting and consuming traditional foods. The popularity of nature and culture documentaries from the Arctic shows that people outside the region value the existence of Arctic peoples, landscapes and wildlife without gaining tangible benefits from them. Regulating and supporting services from the Arctic are also important at a global scale, if one considers the huge role of Arctic sea ice and permafrost in climate regulation and carbon storage. The different scales at which the Arctic provides ecosystem services is represented in Figure 13.

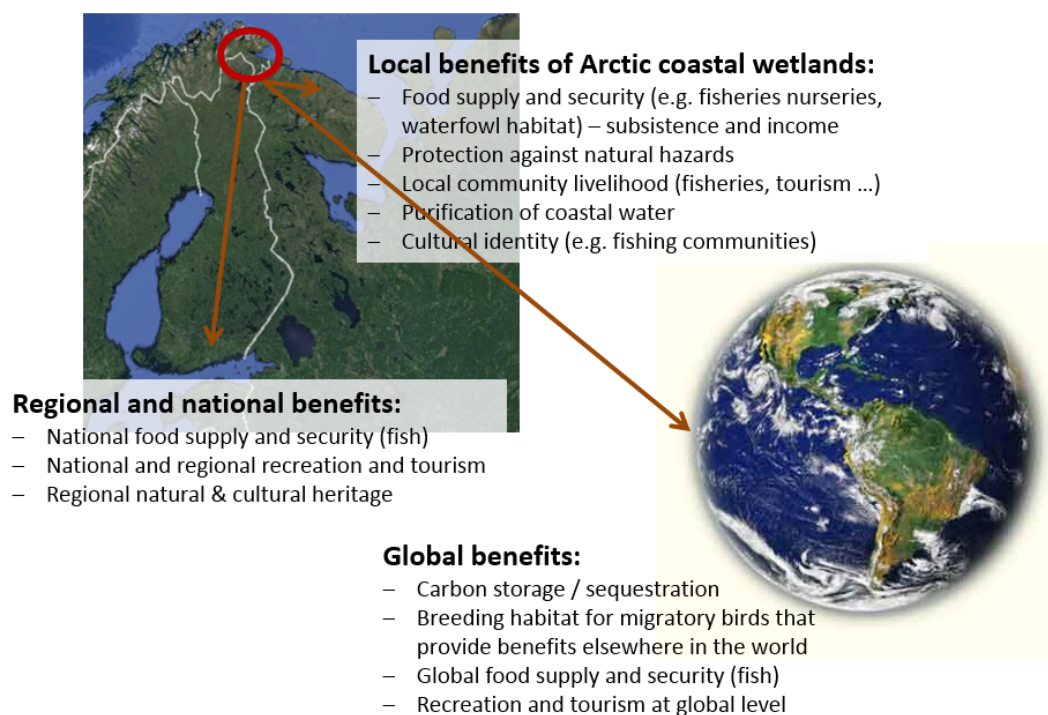


Figure 13 Illustration of the flow of ecosystem services from an Arctic ecosystem (coastal wetland) to a range of beneficiaries at different scales (source: M. Kettunen)

Recognizing, and communicating this vast symphony of cultural and natural capital of the Arctic is crucial in informing decision makers about the impacts of developments and management changes, and about potential trade-offs. At the same time, in a region of global importance and comprised of eight nations and numerous cultural groups and indigenous peoples, finding consensus on the appropriate balance between sometimes conflicting values is extremely challenging. While this task is challenging, it is also urgent to recognize and demonstrate especially the non-monetary values of the Arctic. The fast pace for change and development in the Arctic means decisions are already being taken that will have immense local and global consequences. These decisions should be informed by an adequate representation of Arctic values.

An international study¹⁷ currently underway aims to scope out how arctic biodiversity objectives, values and ecosystem services can be incorporated into decision-making and support ecosystem based management. Preliminary insights gained so far emphasize that any valuation effort needs to be based on integrated and commonly accepted frameworks, must be open to diverse perspectives and responses, pay special attention to cultural values and indigenous views and knowledge, and highlight human health values as those are often overlooked. While we will never be able to conclude on the value of the Arctic, it is clear that the Arctic is invaluable to its people and the planet. Any attempt to describe and recognize its value needs to encompass diverse (value) perspectives and has to make sure that especially non-monetary benefits are adequately reflected.

5.7 Protecting the Coral Triangle to secure food and livelihoods

As the world's centre of marine life on the planet, the Coral Triangle's natural wealth directly sustains more than 130 million people living along the coasts of this 6 million square-kilometre ocean expanse in Asia-Pacific. The annual estimated retail value of the trade in live reef food fish, one of the most lucrative and distinctive of the region's reef-based fisheries is USD 1 billion (Warren-Rhodes *et al.* 2003, Sadovy *et al.* 2003). Indonesia, the Philippines and Papua New Guinea are among the top 10 tuna producing countries in the world. The value of tuna exports from these three countries, plus Malaysia and Solomon Islands, is estimated to be close to USD 1 billion (FAO FIGIS, 2011). The annual value of nature-based tourism in the Coral Triangle is estimated to be worth USD 12 billion (PATA, 2012). All these benefits rely on healthy coastal and marine habitats through the effective protection and management of key areas that are vital for people's food security, livelihoods, and economic stability.

To ensure that this region's natural capital is safeguarded, the governments of the six countries in the Coral Triangle – Indonesia, Malaysia, Papua New Guinea, Philippines, Solomon Islands and Timor-Leste – came together in 2007 to form a multilateral partnership now known as the Coral Triangle Initiative on Coral Reefs, Fisheries and Food Security (CTI-CFF). The CTI-CFF is an example of a regional framework under which governments, private sector, civil society, donors, and development partners collectively aim for the sustainable management of coastal and marine resources in the region. In 2012, the CTI-CFF endorsed a Coral Triangle Marine Protected Area System (CTMPAS) Framework and Action Plan, which contains criteria for the effective management of MPAs and guides the development of a system of MPAs that addresses multiple issues including biodiversity conservation, fisheries management, and climate change adaptation. The CTMPAS is a system of prioritized individual MPAs and networks of MPAs that are connected, resilient, and sustainably financed. These MPAs and networks are designed to be able to generate significant income, livelihoods, and food security benefits for coastal communities, as well as to conserve the region's rich biological diversity.

In 2014, the Tubbataha Reefs Natural Park in the Sulu Sea, in the Philippines was identified as a flagship site for the CTMPAS. Established in 1988, this 970 square-

¹⁷ The TEEB Arctic scoping study is being developed by the Conservation of Arctic Flora and Fauna working group (CAFF), with Sweden as the lead country, jointly with partners: the UNEP TEEB Office, the UNEP Regional Office for Europe, WWF Global Arctic Programme and GRID-Arendal.

kilometre area is a nationally designated no-take MPA – the largest in the Philippines. Governance incentives have made Tubbataha successful over the years which include:

- **Green marketing** of products and services from the MPA through dive tourism - generating \$80,000 to \$110,000 a year from conservation fees which support park management, local community livelihoods, local infrastructure and the improvement of public facilities.
- **Economic compensation** for foregone profits to restricted users (i.e., local fishers)
- Public **communication, education, and awareness** raising activities;
- Alignment with international, regional, national, and local **regulatory obligations** that require effective MPA conservation and an effective judicial system;
- **Participative governance** structures and processes; transparent participation and decision-making processes; and
- **Equity and stewardship** strategies that imply sharing of tourism revenues as a compensatory mechanism and co-ownership of the vision to conserve Tubbataha and take pride in it (Dygico *et al.*, 2013)

Monitoring for the last 15 years in Tubbataha has shown that the live coral cover has been stable at 45-50% after the bleaching of 1998, when coral cover declined by about 22%. Fish biomass, similarly, fluctuates on a yearly basis but has an increasing trend and for the last decade has remained stable at 200 Mt/sq km, which is four times the fish biomass of the average healthy reef in the country. Commercially important species are growing into maturity thus indicating that spawners are protected in the site. This seeds the fisheries in the greater Sulu Sea where the Park is located.

Good management requires good knowledge on the state of the MPA. Therefore, a Management Effectiveness Tool has been developed in the Philippines, which measures the effectiveness of a number of MPAs. Figure 14 shows the positive results of the assessment for Tubbataha.

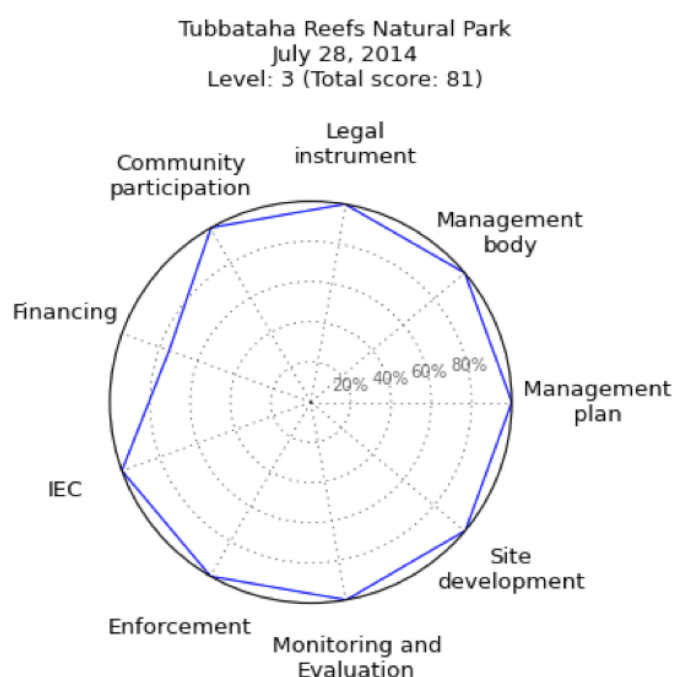


Figure 14 Management effectiveness of Tubbataha Reefs Natural Park, The Philippines (Source: <http://www.mpa.msi.upd.edu.ph/meat/>)

6 Caveats and limitations

The analysis is characterised by substantial uncertainties and is important to recognise that the analysis involves an aggregation of uncertainties from multiple sources (data, functional relationships, parameter values) at each step. The results are currently presented as single point values, which do not reflect the large ranges within which these values are likely to fall. Ideally we would be able to provide a quantified measure that reflects all sources of uncertainty in the analysis, e.g. in the form of an interval around the predicted value and the probability that the true value falls within that range. Computing such prediction intervals, however, requires information on the precision of the data and models used at each stage in the analysis. In the absence of such information, the following caveats and limitation provide a descriptive assessment of the robustness of the results.

1. The scale of the analysis in this study is global and necessarily involves large generalisations. Although the analysis is performed at the scale of individual MPAs and ecosystems, the disaggregated results for individual MPAs are unlikely to be valid and should not be used for decision making regarding the establishment or expansion of MPAs at specific locations. The results of this study do not imply that all MPAs are economically viable. The net benefits from potential MPAs are likely to span a wide range, including negative returns. Careful work is required to consider the circumstances of each proposal, and the social, economic and environmental considerations prevailing there. Similarly, economic evaluations of MPA networks should be conducted at the network scale. In each case there is a need for critical and objective evaluation of MPA implementation and effectiveness (Hargreaves-Allen *et al.*, 2011). In developing new MPAs, full use should be made of existing knowledge and resources for designing effective MPAs (e.g. Salm *et al.*, 2000; Roberts *et al.*, 2003; IUCN, 2008; McLeod *et al.*, 2008, Hargreaves-Allen *et al.*, 2008).
2. The analysis is limited in terms of its coverage of both costs and benefits associated with the expansion of MPAs. On the cost side, we are unable to quantify and value the full set of opportunity costs resulting from MPA expansion. These include costs to shipping; oil, gas and mineral extraction; off-shore wind power generation; and subsistence fishing. On the benefit side, we are unable to quantify impacts to all ecosystems (e.g. seamounts, seagrass, kelp forests) and all ecosystem services (e.g. existence values associated with marine biodiversity) that are potentially positively impacted by MPAs. We therefore recognise that our analysis only provides a partial assessment of all costs and benefits and should be revisited as the necessary data and knowledge become available.
3. The marine ecosystems for which we are able to model the benefits of MPA coverage are predominantly coastal and tropical (i.e. coral reefs, mangroves and coastal wetlands). It has proved harder to model the effects of MPAs on ocean and temperate ecosystems and polar regions are omitted from the analysis due to issues of data quality.
4. The estimation of establishment and operational costs of an MPA are modelled only as a function of MPA size using published cost functions (McCrea-Strub *et al.*, 2011; Balmford *et al.*, 2004). As such, the estimated costs do not reflect other potentially important determinants of cost, such as capital and labour costs, distance from ports, scale and proximity of other MPAs in the vicinity (that would reflect potential economies of scale across networks of MPAs). The Balmford *et al.*

(2004) study reports the results of alternative specifications of the cost function that includes distance to nearest inhabited land and the Purchasing Power Parity (PPP) index value of the country. The estimated coefficients on these variables, however, are counter to expectations (i.e., costs decrease with both distance and PPP). We therefore decided to use the simple cost function specification that includes only the size of MPA.

5. The exploratory scenarios are defined by a small set of simple rules in order to explore broad alternative strategies for MPA expansion. The spatial allocation of MPAs under each scenario does not therefore reflect the wide range factors that would ideally be considered in the actual siting and design of an MPA. It is also important to stress that none of the exploratory scenarios represent an economically optimal allocation of MPAs.
6. The “protect to preserve” (P2P) scenario allocates MPAs to areas with high biodiversity and low human impact. It is intended represent a strategy of using MPAs to avoid potential future damage to areas of high biodiversity. The impact of an MPA in such a location depends in part on what would happen in the absence of the MPA. If no damaging activities would take place anyway, the MPA is essentially redundant. On the other hand, if damaging activities would take place in the future, the MPA may play an important role in protecting the marine ecosystem. In short, the full assessment of this exploratory scenario requires a baseline scenario that fully describes the presence or absence of future damaging activities. This is beyond the scope of the present study.
7. The nature of the methodology used for assessing the benefits of expanding MPA coverage is likely to produce higher returns for the preservation of marine ecosystems than for restoration. This is because we are better able to quantify the extent of avoided losses than the extent of restoration under MPA designation. The results will therefore tend to favour scenarios that target preservation over mitigation of existing pressure. For this reason, the assessed benefits for the P2P scenario may be greater than the P2M scenario.
8. The maximum sizes of MPAs in our set of scenarios greatly exceed the maximum size of existing MPAs and also the maximum size of MPAs that have been assessed and used to estimate management cost functions. The size of the largest MPA included in the data underlying the Balmford *et al.* (2004) cost function is 344,000 km², whereas the largest MPAs in our scenarios exceed 2 million km². In using this cost function to estimate operation costs, we are extrapolating beyond the range of the underlying data. In practical terms, this means that the estimated operation costs per km² can be very low for very large MPAs. To address this issue we limit the lower bound of operation costs per km² to be equal to the minimum cost in the data used by Balmford *et al.* (2004), i.e. US\$ 4.30 per km².
9. In the underlying biodiversity data, “seamounts” are not directly taken into account (habitat in general is not), however, one of the parameters used to model the distribution of species in the biodiversity map, is depth. The underlying biodiversity map is modelled on the physical parameters under which a species exists, among those parameters is depth. So If a species encounters an habitable depth (seamount) and that seamount is within the species mobility range (measured from actual species sightings) and if all other parameters are suitable too, it is likely to encounter biodiversity hotspots on seamounts. In general, the biodiversity map resembles the habitat map to a large extent. Regarding our scenarios, seamounts are likely to be prioritized in MPA allocation (assuming that

their spatial extent is significant enough to survive the depixelation), except for under the easy-to-expand (E2E) scenario, which targets areas of low biodiversity.

10. The analysis does not take account of potential displacement effects of protected areas. Restricting human activities within MPAs may, to some extent, lead to the displacement of those activities to unprotected areas, which may experience greater degradation and loss of ecosystem services as a result. We are not able to quantify and value this cost of expanding MPAs.
11. The potentially positive spill-over effects of MPAs on commercial fisheries are assessed through arbitrary assumptions regarding the scale of spill-over effects and the baseline rate of decline in the value of fisheries. No conclusive spill-over effect from MPAs was identified in the literature review. To some extent, a positive effect of MPAs on fisheries is included in the assessment of benefits from protected coral reefs, mangroves and wetlands since the value of related fisheries are included in the value functions for those biomes.
12. Assessing the costs or benefits to fisheries of expanding MPA coverage requires a complete understanding of what would happen in the absence of MPA expansion (i.e. the baseline scenario). This includes a description of the resource over time (i.e. whether it is declining, stable or increasing) and also a description of how the fishery would be managed. There are a number of pre-existing legal obligations for nations to manage fisheries sustainably and protect biodiversity (e.g. the UN Fish Stocks Agreement) that would require changes in fishing practices and effort. The introduction of other fisheries management instruments is not taken into account in our analysis. Doing so would affect both the estimated costs and benefits to fisheries of MPA expansion.
13. The time horizon of the analysis is 2050 and we have attempted to account for changes in relevant environmental parameters up to that point. The impacts of climate change and ocean acidification on marine ecosystems are, however, expected to increase markedly beyond 2050. The benefits of more action now to protect and build ocean resilience in the face of climate change and ocean acidification will therefore only be realized in the long term. Our analysis does not measure these benefits but they provide a further argument for current expansion of MPAs. Climate change impacts might also have profound implications for the design and effectiveness of any marine management measures (including MPAs). For example, it may be necessary to develop “mobile MPAs” to protect spatially dynamic spawning areas.

7 Conclusions and recommendations

This study develops a set of six mapped scenarios for the global expansion of MPAs. The scenarios vary along two dimensions: 1. the total coverage of MPAs as a proportion of EEZs, ABNJs and key marine habitats; 2. the characteristics of target locations in terms of biodiversity and extent of human impact. We conduct an economic assessment of these scenarios by estimating and comparing the costs and benefits of each scenario. Where feasible the analysis is conducted at a high spatial resolution, allowing the estimated costs and benefits to reflect characteristics and context of each MPA. The results of this cost-benefit analysis show that all six scenarios are economically advisable (the ratios of benefits to costs are in the range 3.17 – 19.77). In the case of the scenario that achieves 10% coverage of total marine area and targets areas with high biodiversity and low human impact, each dollar invested yields a return of around 20 dollars in benefits. On this evidence the expansion of MPA coverage can be recommended from an economic perspective.

It is important to stress, however, that this analysis is highly constrained by the available data on marine ecosystems, services, costs and benefits; and by the available knowledge on how MPAs affect ecosystems and the provision of services. The paucity of understanding of the relationships between MPAs, ecosystem function, ecosystem services, and the value of services is widely documented (e.g. Potts *et al.*, 2014) but needs to be addressed in order to produce more complete analyses of the human welfare effects of expanding MPAs. This means that further, well-designed, research is required to specifically quantify the linkages between MPAs and economic values. Among the many aspects that require more in-depth understanding, we note here the need for further research on a number of key issues.

In addition to building a more complete assessment of all relevant costs and benefits from MPA expansion, future studies should also assess the distribution of costs and benefits across different stakeholder groups and possibly countries. In the present study, the assessed costs are incurred by governments (establishment and operating costs) and the commercial fishing sector; whereas the assessed benefits primarily accrue to coastal communities benefiting from coastal protection, reef and mangrove related fisheries, recreation and tourism.

Developing a fuller understanding of the opportunity costs of MPAs for maritime industries appears to be tractable and offers a starting point for extending the current analysis. These costs include the impacts of MPAs on oil, gas and mineral extraction; off-shore wind power generation; and subsistence as well as commercial fishing. This would be useful information in any discussion with impacted sectors on the potential costs involved.

There is substantial and growing evidence of the positive effects of MPAs on populations of individual organisms and ecosystems. In order to use this evidence to model MPA effectiveness, further understanding of how specific MPA parameters such as age, size, protection level and location is required.

The reviewed evidence on the effectiveness of MPAs in enhancing the provision of ecosystem services is weak. The ability to understand the relationships that exist in different contexts between MPA designation and ecosystem service provision would be greatly improved in the future by research that employs study designs capable of isolating and quantitatively measuring the biophysical impacts of MPAs, as well as by research that endeavours to directly measure ecosystem service provision through time. As studies of this nature increase in frequency, so too will the ability of

researchers to understand the nature of ecosystem service provision, and by extension, also the economic valuation of MPA designation.

Regarding recommendations for future assessments of MPA expansion, the scenarios that are assessed could be developed in several directions. New scenarios could apply alternative criteria for targeting MPA locations, such as ecological uniqueness, coastal community dependence on marine resources, or fish spawning sites. Potential network effects could also be taken into account in modelling the location of future MPAs. This would greatly increase the complexity of the data and models for scenario development but also provide a more complete picture of where to spatially allocate MPAs. Scenarios could also be developed to describe alternative combinations of no-take and multi-use MPAs. This would provide a more realistic case in which MPAs enforce a mix of both spatial and temporal restrictions. Modelling the effects of multi-use MPAs would require a nuanced understanding of how different levels of protection impact the provision of ecosystem services.

Current lack of knowledge should not, however, limit action on marine management and conservation. Taking action provides a 'no regrets' option, which means that a more informed use of marine resources can be made in the future. It is recognised that MPAs cannot address the whole gamut of threats facing marine ecosystems (Kearney and Farebrother, 2014). An integrated approach to marine management is therefore needed. The focus on MPAs as a key policy instrument for marine management should be kept in perspective and used in conjunction with other policy instruments. The adoption of MPAs should not become an excuse for not implementing other recommended management measures. MPAs are an essential element of the "management tools mosaic" but should not be treated as the panacea. In practice MPAs should be used to reinforce and strengthen other forms of management and complement other types of intervention.

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Appendix A Scenarios for expansion of Marine Protected Areas

Introduction

Over the past decade several policy targets have been set for the expansion of Marine Protected Areas (MPAs) in the future. These targets are elaborate in their description but also include terms open for interpretation (CBD, 2012). Meeting these policy targets can therefore be achieved in several ways.

The aim of this work described in this Appendix is to map a set of scenarios for expanding the coverage of MPAs by conducting a simplified marine spatial planning exercise on a global scale. These scenarios can aid in visualizing MPA expansion and act as a source for debate on how our future oceans will look. The exploratory scenarios developed in this study are loosely related to the CBD Aichi Target 11 on Protected Areas and on a similar target for protecting 30 percent as referred to in the Durban Target and the Promise of Sydney. It is not the intention, however, that the scenarios developed here address all aspects of the CBD or Durban targets. Aichi Target 11, adopted in 2010 at the 10th Conference of the Parties in Nagoya, Japan, requires that:

By 2020, at least 17 per cent of terrestrial and inland water, and 10 percent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes. (CBD, 2010)

Initially, these several aspects such as, representativeness and connectivity of these targets will be further analysed and considered in the marine spatial planning model. Furthermore, an analysis of the jurisdictional areas will be made as this will act as a framework for the scale at which the criteria are met.

Designing future scenarios for Marine Protected Areas

There are several routes in achieving a higher level of MPA coverage. Although the policy targets do provide some rationale on this, a lot is still left open. The establishment of MPAs is mostly difficult because of conflicts of interest in (economic) activities in the potentially conserved areas. The recent establishment of large MPAs in remote areas has been criticized and considered as easy MPA establishment to meet policy targets (Leenhardt, Cazalet, Salvat, Claudet, & Feral, 2013). Future MPAs in areas of importance for human activities is expected to be more difficult due to conflicts of interest. In this study, global cumulative human impact is therefore considered as a proxy to identify areas of current importance for human activities. Secondly, the presence of biodiversity is also of importance. Biodiversity is one of the key ecological characteristics to be preserved under the policy targets. Both aspects are combined in a matrix generating four scenarios with different priorities for conservation.

Table A.1 Scenarios for MPA expansion based on biodiversity and human impact

	Low Human Impact	Low Human Impact
Low Biodiversity	Easy to Expand -E2E- <i>Low Biodiverse impact with low human interference</i>	
High Biodiversity	Protect to Preserve -P2P- <i>High Biodiverse areas away from human impact. Protection preserves the area from potential future impact.</i>	Protect to Mitigate -P2M- <i>High Biodiverse areas under High Human Impact: Protection is mitigating the Impact</i>

The scenario that targets MPA allocation in high biodiversity areas with high human impact can be seen as a scenario for mitigating the current human pressures on biodiversity and hence is titled “Protect to Mitigate” (P2M). The scenario “Protect to Preserve” (P2P) has a preference for high biodiversity areas away from human impact and can therefore be regarded as a scenario to preserve areas that are not under high impact yet. The low biodiversity scenarios do not represent any policy target, but were generated for comparing economic valuation of these scenarios. The “Easy to Expand” (E2E) scenario, in which areas with low biodiversity that are not of interest for human activity are targeted, does not represent a policy goal and can be considered as a scenario that attempts to meet the required areal extent of MPA coverage at low cost (Leenhardt, Cazalet, Salvat, Claudet, & Feral, 2013).

Generation of the conservation priority maps

The above-described scenarios were partly selected due to considerations of data availability. For human impact, there is global data compiled (Halpern, *et al.*, 2008). Biodiversity data on a global scale is not available but modelled data on the worldwide distribution of biodiversity is used instead. For biodiversity, the all species biodiversity map from AquaMaps (Kaschner, *et al.*, 2013) was used. Both datasets are high resolution, allowing the generation of the scenarios on a 10km pixel resolution. Three priority maps were generated from this data in order to determine areas of preference for MPA allocation.

Both datasets were reclassified using the Jenks natural breaks classification method in order to allow equally weighted overlaying. In the low biodiversity and low human impact parts, the inverse values were used in the equally weighted overlay. Subsequently, a low pass filter to exclude loose pixels was used and focal averaging to avoid highly pixelated priority maps, which in turn would lead to small pixelated MPAs after the model run.

As a result, the four priority maps were delivered, with the following caveats:

- The depixelation is a step of generalization. Local and regional variation is smoothed out. Multiple small MPAs in one region are then aggregated to a smaller set of larger MPAs. In smaller planning areas, this could lead to not extracting exactly 10% or 30% of the area due to rounding issues.
- The input data is seen as global, however, many of the underlying input data for the Arctic and Antarctic data are less reliable. Therefore, only areas up to 70 degrees North and South are considered in the analysis. The study is therefore limited to this geographical extent.

Incorporating policy targets in the planning model

Policy targets are not only percentages of marine cover and in addition to biodiversity they include some MPA design properties. Three of the key aspects are area, connectivity and representativeness. Running analyses on a global scale has limitations on how completely such criteria can be assessed. Data availability and processing time are the main constraints in developing the future scenarios. This section discusses how the design properties were assessed and incorporated into the spatial planning model.

Area

Area of MPA coverage, expressed as percentage of the global marine environment, is one of the key criteria of targets used in discussions and communications by environmental organizations. The CBD Aichi Target 11 requires a global marine coverage of 10%. More long-term and ambitious goals focus on 30%. This study therefore focusses on the expansion of MPAs to reach up to 10% and 30% global marine coverage. The final scenarios will therefore all have two variants where there is respectively 10% and 30% global coverage of MPAs.

Connectivity

As opposed to area, connectivity is more open to ambiguous interpretation. In terms of the CBD target, this is specified in more detail, describing connectivity as *“linkages between sites in a network created through larval dispersal, migration of organisms and the mixing of waters through currents and other oceanic physical processes”*. As the scenario development is on a global scale and consists of an extreme variety of ecosystems and habitats and will require a very large amount of processing time as well as a wealth of data, it will therefore be difficult to cover this aspect in full. In addition, the valuation functions that are used to assess the scenarios do not explicitly take connectivity of MPAs into account. Specific allocation rules with regard to connectivity will therefore not be included in the model. However, the selected biodiversity model to generate the scenario does take dispersal and migration into account to some degree.

Representativeness

According to the CBD Targets, MPAs should be developed in representative networks. Representativeness of an MPA network requires that all habitat types, species and characteristic species communities, as well as areas for important life stages such as spawning, breeding and migration sites are included. Apart from migration, most of the above described criteria can be found in key habitat types. For limited computation

times and limited data needs, only these key marine habitat types were taken into account with regards to representativeness. These habitat types also have data available for the valuation of ecosystem services, making them an essential part of the analysis. The following habitat types were taken into account in the marine spatial planning model:

1. Mangroves (Giri, *et al.*, 2011)
2. Coral Reefs (UNEP-WCMC, 2010)
3. Sea grass (UNEP-WCMC, 2005)
4. Coastal Wetlands (WWF, 2004)

Although this is a very limited selection, these habitat types are very well documented and have nearly worldwide data availability. These habitats do not therefore fully cover representativeness, but provide an indication. It is important to note that the biodiversity data indirectly includes other habitats in which there is high biodiversity, including migratory routes.

Establishing planning units based on ocean jurisdictions

Planning units are integral parts of a marine spatial planning exercise. Planning units are defined units of space that can be indicated as MPA or not in the model. The assumption is made that all ocean jurisdictions should comply with the set of criteria in themselves. This avoids the outcome that countries with high marine biodiversity establish proportionately higher MPA coverage than other countries. The same equal share principle is applied for areas beyond national jurisdiction (ABNJ).

Exclusive Economic Zones (EEZs) are the only areas considered as national jurisdiction, as the status of the extended continental shelf (ECS) is not seen as an area of jurisdiction due to its status being mainly submissions. Areas beyond national jurisdiction don't have specific governing body, making it difficult to divide this area. However, to ensure a more equal global spread in the high seas, the FAO fishing areas were used as jurisdictional units for the high seas. The resulting jurisdictions map *combined* the EEZs (VLIZ, 2014) with the FAO Areas (FAO, 2014).

Running the model

Existing MPAs were kept protected in the future scenarios. As a baseline, the database version of October 2014 was used (UNEP-WCMC, 2014). This database corresponds with the most recent publication on global MPA coverage, totalling 3.4% worldwide. Additional MPAs under each scenario are allocated using the respective conservation priority maps and ensuring equal proportionate distribution per jurisdictional zone and per habitat.

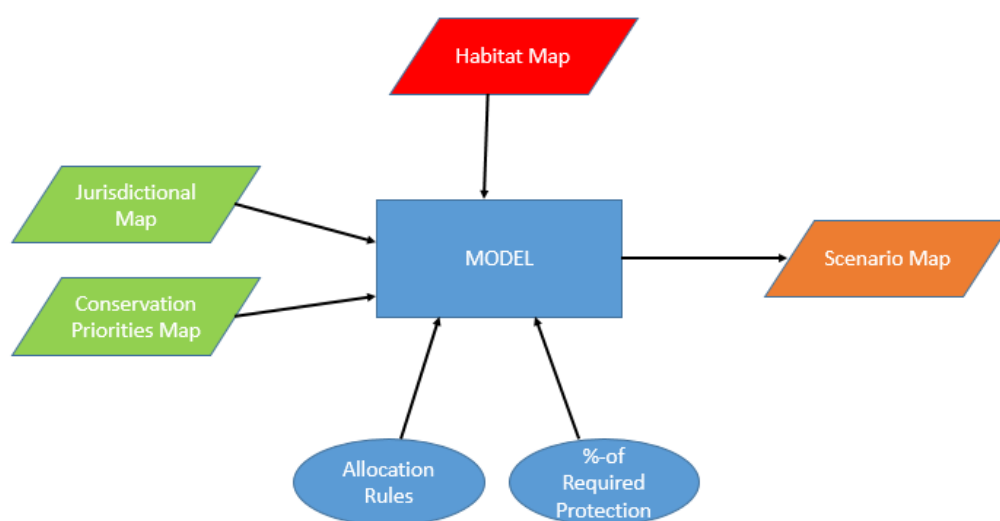


Figure A.1 Scenario generation model

The model uses three iterations for adding existing MPAs, expanding MPA coverage across key habitats and allocating the remaining targeted proportion of area.

Iteration 1: Adding Existing MPAs

Existing MPAs are added to every jurisdictional unit and subsequently the percentage of protection is calculated and the remainder that needs to be covered. If a jurisdictional unit reaches the desired degree of allocation, the planning unit is complete and will not be further analysed in following iterations. The area of coverage is the prime indicator for this, so other criteria such as biodiversity or human impact are not necessarily met. It is potentially the case that the planning unit is only compliant with the required percentage of protection.

Iteration 2: Reviewing Habitat

All habitat pixels in the planning unit are designated a priority value according to the conservation priority scenario. The respective top 10 or 30 percent of cells are allocated to MPAs to ensure a limited degree of representativeness. Subsequently, a recalculation of the degree of protection is made. If the degree of coverage is achieved in the planning unit after this iteration, the zone will not be further assessed in the next iteration.

Iteration 3: Remaining Area

After iteration 2, the remainder of non-protected pixels will be reassessed in order to ensure that the required proportion of MPA cover is reached. It should be noted that in small planning units and areas with limited variation in conservation priority, rounding errors can occur causing a small deviation of the desired target.

Summary of model caveats

A perfect spatial planning exercise on a global level in limited calculation times is not possible. The main assumptions and caveats for the scenario development are summarized here:

- Existing MPA allocation is not reviewed. Planning units that already meet the required degree of protection may not meet the other criteria defining each scenario.
- Smoothing the results has led to generalization and a slight decrease in variety in planning units. Causing small deviations from the policy targets but were assessed insignificant
- The quality of data for areas beyond 70 degrees North and South was too limited and therefore this region was excluded from the modelling exercise
- The modelling exercise uses a 10km raster based approach. The raster projection was based on Mollweide projection, which causes slight differences in area calculation at the higher latitudes.
- Connectivity and representativeness are modelled to a limited extent due data and computational limitations.

Results

Per scenario, maps were generated with the allocation of future MPAs. The mapped scenarios are not presented in this report due to concerns that they could be misinterpreted as spatially explicit recommendations for the siting of MPAs. The development and siting of specific MPAs and networks of course requires a rigorous process of research, consultation and assessment that reflects multiple factors relevant to each case. The scenarios developed in this study do not replicate that process and should only be used for the purpose in which they are intended: the exploratory assessment of the potential global net benefits of expanding MPA coverage.

General observations for all scenarios

Planning units that already comply with 10 or 30 percent MPA coverage remain the same in all scenarios. EEZs that already have high levels of protection, such as Australia, show little difference between scenarios for 10 percent coverage and limited differences between scenarios for up to 30 percent.

Abrupt edges and lines can be seen in the result. This can be explained by the shape of some planning units such as the FAO fishery areas, EEZ boundaries or existing MPAs.

When increasing the proportion of MPA coverage from 10 to 30 percent there is a tendency to expand existing MPAs rather than establish new MPAs. The shapes of the MPAs also become more complex under 30% coverage as compared to 10%.

Protect to Mitigate (P2M)

The Protect to Mitigate scenario is influenced by the gradient in conservation priority, which decreases when moving away from shore. Most coastal areas are highly biodiverse but also under high impact. The allocation rules tend to create groups of MPAs along the coast in the EEZ, and as a corollary there is no protection in the EEZ

further from the coast. The areas beyond national jurisdiction then sees a continuation protection from the EEZ boundary, resulting in corridors of non-protection between coastal MPAs and the boundary of EEZs.

Protect to Preserve (P2P)

In the protect-to-preserve scenario, protection takes place further away from shore. Within EEZs the protected areas are distributed to protect key habitats, and generally habitats further away from shore. A similar pattern can be observed in areas further away from cities and ports as these are generally areas of lower impact.

Easy to Expand (E2E)

The Easy-to-Expand scenario focuses on areas with low biodiversity and low human impact. Most typically, these are in the centre of open oceans just North and South of the inter-tropical convergence zone (ITCZ). Some exceptions to this are remote coasts, such as desert coasts.

When looking at current allocation of remote high-seas MPAs, it can be seen that this type corresponds most closely to the E2E scenario. For example, the Charlie-Gibbs fracture zone MPA in the northern Atlantic established by OSPAR.

Discussion

Running models on a global scale using a holistic approach including a lot of variables is a complex process. The goal of this scenario develop process is not to predict the future, but to examine plausible futures. This scenario study gives an indication of future scenarios and deals with important aspects of current policy discussions. It visualizes how the future oceans could look with a higher degree of protection.

The scenario generation could, however, be improved by including more information on connectivity and representativeness. This is now covered to a very limited extent through the biodiversity data and the habitat coverage criteria.

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Appendix B The effects of MPAs on organisms and ecosystems: A review of the literature

Introduction

This appendix provides an overview of the empirical evidence on how organisms and ecosystems are affected by marine protected areas (MPAs). Information for this review is derived from scientific articles and government publications that present quantitative findings concerning the biological and ecological effects of MPAs. Where available, published meta-analysis of literature is used in order to draw on existing syntheses. These studies compare spatially and temporally variable parameters inside an MPA to its surroundings. Such comparisons highlight the differences between protected and non-protected areas and have been made for a wide range of species (fish, corals, mammals etc.) and for different parameters (abundance, density, weight, diversity etc.). Most analyses of the impacts of MPAs can be described by the following three categories:

1. *Spatial* data on population demographics *within* and *outside* of an MPA. The premise is that positive biological parameters (e.g. abundance, density) are higher within an MPA because the number and/or magnitude of stressors are lower.
2. *Temporal* and *Spatial* data on population demographics *within* an MPA, *before* and *after* implementation of the MPA. The premise is that positive biological parameters within the MPA are higher ex post establishment of the MPA when the number and/or magnitude of stressors have been reduced.
3. *Temporal* and *Spatial* data on population demographics *outside* an MPA, *before* and *after* implementation of the MPA. This data is used to measure possible spillover effects of MPAs. If population densities are higher *outside* the MPA *after* it has been implemented, then this could be evidence of a positive spillover occurring.

Ideally, the assessment of an MPA would cover all three of these types of analysis and associated effects. Most often, however, this is the exception rather than the rule.

Making any generalized assessment of ecological changes associated with MPAs is a challenging task. The marine ecosystems in which MPAs are implemented are complex and highly diverse in terms of habitat types, ecological structures and processes, and external stressors. In order to provide an overview and present the information in a structured manner, this Appendix is divided into the following sections: fish and invertebrates; mega fauna; Coral Reefs; and Ecosystem Resilience. This division is based on the common life history traits of the organisms and associated similarity of impacts of MPAs, and for practical reasons pertaining to the accessibility of data on the subject. Although responses to the implementation of an MPA are highly variable¹, species sharing similar life history traits are likely to respond in a similar manner^{2,3}. It is for this reason that fish and invertebrate species are grouped together. Elasmobranch species, i.e. shark species, have been excluded from this category, however, and included in the section on mega fauna species. These organisms are characterized by being large, slow growing and having vast home ranges hereby sharing similar traits with species such as dolphins and sea turtles. In the final section primary producers such as seagrass, corals and mangroves are discussed. How these organisms are affected by protected areas and

how this in turn influences the resilience of the ecosystem is the focus of the final section.

Fish and Invertebrates

Global fisheries catches have been in decline since the last two decades of the 20th century, professing the finite state of some of our oceans resources⁴⁻⁶. In large, the decline of fish and invertebrate populations can be attributed to factors such as; overfishing, habitat destruction, invasive species, climate change and pollution⁷. Each of these stressors has, to varying degrees, the capacity to negatively impact the marine environment. A large number of MPAs established to date, have the goal to replenish fished populations⁸. How successful MPAs are in achieving this goal is still under debate⁹. Despite a drastic increase in studies relating to MPAs, the complexity of how these processes (stressors, population dynamics, and protection within specified areas) interact and their cumulative impact remains difficult to quantify¹⁰. This section of the report falls within this context and aims to shed light on potential effects of MPAs on fish and invertebrate species.

Declining populations of fish and invertebrates can lead to substantial and irreversible changes in ecosystem structure and functioning¹¹. Marine protected areas play a role in preventing the decline of fish and invertebrate populations. Although MPAs are not a panacea, they do have the potential to preserve or restore habitats and species.

Marine protected areas can create beneficial effects for the ecosystem through a diverse set of ecological pathways. A decrease in fishing effort can directly benefit fish populations. There are also indirect pathways by which fish or invertebrate species can benefit from an MPA. For example, the complete removal of fishing activities within an MPA can help restore benthic communities. Prohibiting trawling and dredging would allow for the recovery of habitat complexity, benefitting survival rates of juvenile fish and promoting species diversity¹². MPAs could also have a positive effect on migratory species because of a shift occurrence in habitat use. Such species might remain longer within the MPA as conditions are more favourable compared to outside. The longer they remain in the MPA the fewer anthropogenic stressors they encounter.

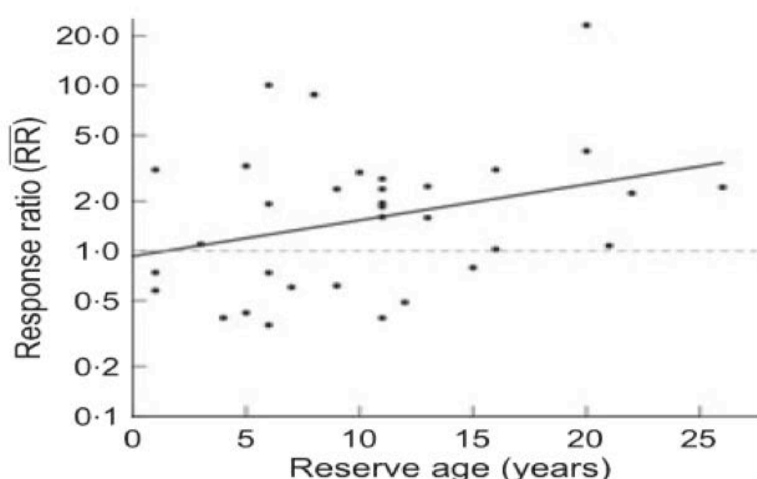


Figure B.1 The relationship between age of an MPA and the ratio of fish density within and outside the MPA. The striped line ($RR = 1$) represent when fish density outside the MPA is equal to fish density within the MPA. A value greater than $RR=1$ indicates that there is a higher density of fish within the MPA. Source: Molloy, McLean, and Côté (2009)

In order to provide guidance on how MPAs should be designed and managed it is of interest to observe whether certain characteristics of an MPA (e.g. age, size, protection level) can affect the outcome of conservation efforts. The extent to which scientific literature provides evidence on the conservation outcomes of different MPA characteristics, however, is limited.

Regarding the temporal scale over which MPAs can be observed to have positive effects on biological parameters, there is some evidence. Figure 1 shows the relationship between MPA age and the ratio of fish density within and outside the MPA. It is observed that this ratio increases by approximately 5% for each year following MPA establishment¹³. Some research findings report stronger effects, with fish densities in the range of 5-14 times higher in MPAs over 10 years of age as compared to fish densities in younger MPAs¹⁴. In addition, older MPAs (>10 years) have been observed to have significantly larger sizes of fish, which on average produced 20-100 more eggs compared to their smaller counterparts¹⁴. Another study, conducted by Claudet *et al.* (2008), also indicates that the age of an MPA influences the amount of fish within the MPA boundary¹⁵. Their study on commercially harvested fish states that the age of an MPA partially explains the variability of the effects of protection. As MPAs increase in age, an 8.3% increase in relative fish density is shown. This is a clear indication that age is an important factor influencing the protective capacity of an MPA.

The evidence on the effects of the spatial scale of MPAs on biological parameters shows little positive returns to scale. In relative terms, most studies appear to show no relationship between the percentage increase in fish biomass, density or diversity in and the size of an MPA¹⁶. In absolute terms, larger MPAs will have larger effects than smaller MPAs. However, the study of Claudet *et al.* (2008) is the only study found to show that the size of an MPA does matter¹⁵. The authors state that for every 10-fold increase in size, commercially harvested fish species increased in density by 35%.

The protection level of an MPA has been shown to significantly influence its impact in terms of increasing biomass and density of fish species¹⁷⁻¹⁹. MPAs with a high level of protection have significantly higher biomass and density counts compared to MPAs with intermediate levels of protection. Edgar *et al.* (2010) categorized MPAs into two different levels of protection, high and low, and assessed their relative performance. Their results show that well managed MPAs with high level of enforcement have a significant effect on the presence of carnivorous fish and invertebrate species. In all cases biomass within high-protection MPAs was higher compared to fished zones.

The previous articles only look at single MPA parameters, i.e. size, age or effective management. However the influence of an MPA becomes more evident when comparing multiple MPA parameters simultaneously. MPAs with at least four of the following five key features have been shown to be more effective: age (> 10 years), large (> 100 km²), no take, well enforced, and isolated by deep water or sand²⁰. The more key features an MPA has, the larger the response ratios of fish biomass and species richness. For each additional key feature, the MPA effects on biomass and species richness are increased, reaching a maximum when all five are present. For example, compared to fished areas, MPAs with 3, 4, or 5 key features had 50%, 210% or 350% more fish biomass respectively²⁰.

In general, MPAs have the capacity to increase biological parameters for fish. Lester *et al.* (2009) find in a global synthesis that the impacts on fish biomass, density, species size and species richness increase by 446%, 166%, 28% and 21% respectively⁷ (Table 2). To understand the mechanisms that drive this change, however, requires further research.

Invertebrate species can also benefit from the presence of an MPA. Although variation is large, catches were recorded to be 6-58 times higher inside MPAs compared with outside^{7,21}. Other studies have found that abundance also increased, having density counts that were 12 times higher within reserves and with fecundity rates also improving²². A study conducted on clams showed that after five years, densities were 19 times higher within the reserve and 7 times higher in fished areas compared to the period prior to the establishment of the MPA¹⁴. However, as multiple meta studies have shown, the effects vary between MPAs (Tables B.1 and B.2).

Table B.1 Overview meta-studies on the effects of MPAs on invertebrates. The response ratio is the comparison of biological parameters within and outside or before and after MPA establishment. All are statistically significant. A response ratio of e.g. 1.35 represents a 35% increase within the MPA

Author (year publication)	N	Biological Parameters	Response ratio		MPA Size	
			Mean	S.E.		(km ²)
Halpern ¹⁶ 2003	81	Density	2.04	6.15	Range	0.002-864
		Diversity	1.08	0.17	Mean	44
		Size	0.8	0.22	Median	4
Lester ⁷ 2009	149	Density	2.8	N.A.	Range	N.A
		Biomass	8.77	N.A.	Mean	N.A
		Size	1.26	N.A.	Median	N.A

The evidence on the occurrence of spillover effects for fish and invertebrates is mixed. Spillovers can occur as a result of natural migration patterns, large home ranges, or density dependent processes as the carrying capacity within the MPA is reached^{14, 21-24}. Some studies have shown substantial spillover effect, for example Goñi *et al.* (2009) report a tenfold increase in landings of spiny lobster outside an MPA²¹. However, there is no general trend to be drawn from the literature. García-Rubies *et al.* (2013) find limited evidence of spillovers in a Mediterranean MPA after 19 years of protection²⁷. In addition, spillover effects where they do occur appear to diminish rapidly with distance from the MPA²⁸. Some studies have found that spillovers are no longer evident at distances greater than 2 km from the MPA^{23,28}. This implies that there is maximum distance over which spillover can occur. Fishing dynamics strongly affects this maximum, because in most cases when an MPA is established, fishing effort is displaced to the boundary of an MPA²⁸. Thus, even if spillover is occurring, organisms are exposed to harvest at the boundary of the MPA. This can even result in a net decrease within the MPA as there is no movement back, resulting in a dispersion imbalance²⁹.

Multiple studies have shown that density and biomass is higher within and outside of MPAs post implementation¹³. However it remains difficult to provide conclusive evidence on which factors cause this increase since there are multiple effects that might be confounding the results. Habitats where MPAs are located could be favourable, which might be the main factor that causes an increase in biomass. The level of rugosity of coastal habitat and the latitude both showed correlations with species richness and biomass. Increase in biomass could also be the result of a net increase in immigration and not of an increase in birth, growth or survival rates.

Table B.2 Overview meta-studies on the effects of MPAs on fish. The response ratio is the comparison of biological parameters within and outside or before and after MPA establishment. All are statistically significant. A response ratio of e.g. 1.35 represents a 35% increase within the MPA. N.A.: Indicates either that the data was not present in the article or not significant

Author (year)	N	Location	Biological Parameter	Response Ratio Mean	S.E.	MPA Size (km ²)
Claudet ² 2010	40	Mediterranean	Density	2.46	N.A.	Range
			Biomass	N.A.	N.A.	Mean
		Atlantic	Diversity	N.A.	N.A.	Median
			Size	1.35	N.A.	
Côte ³⁰ 2001	15	Global	Density	1.28	0.16	Range
			Biomass	N.A.	N.A.	Mean
			Diversity	1.1	0.05	Median
			Size	N.A.	N.A.	
Halpern ¹⁶ 2003	81	Global	Density	2.15	2.95	Range
			Biomass	2.943	2.75	Mean
			Diversity	1.713	0.36	Median
			Size	1.353	0.2	
Lester ⁷ 2009	149	Global	Density	3.1	N.A.	Range
			Biomass	4.5	N.A.	Mean
			Diversity	1.25	N.A.	Median
			Size	1.24	N.A.	
Molloy ³¹ 2009	33	Global	Density	1.66	0.21	Range
			Biomass	N.A.	N.A.	Mean
			Diversity	N.A.	N.A.	Median
			Size	N.A.	N.A.	
Sciberras ¹⁸ 2013	62	Global	Density	1.22	N.A.	Range
			Biomass	2.96	N.A.	Mean
			Diversity	N.A.	N.A.	Median
			Size	N.A.	N.A.	
Claudet ¹⁵ 2008	12	Europe	Density	2.46	N.A.	Range
			Biomass	N.A.	N.A.	Mean
			Diversity	N.A.	N.A.	Media
			Size	N.A.	N.A.	

Mega Fauna (mammals, large fish, turtles)

Mega fauna, although being a term that encompasses a broad selection of species (e.g. cetaceans, pinnipeds, sea turtles and sharks), share traits that make protecting them challenging³². For example, most of these species have large home ranges and are migratory. Their large size makes them easy prey to anthropogenic hunting efforts, either directly or as result of by-catch. Maturity is reached relatively late compared to smaller organism and fecundity is low, resulting in slow population growth rates, as is the case for sharks³³⁻³⁵ mammals³⁶⁻³⁹ and turtles^{40,41}. Marine mega fauna are highly threatened by overexploitation and habitat degradation⁴² and many populations have

declined drastically^{32,43,44}. Pollution of coastal waters also results in the demise of mega fauna population, as it affects them directly or indirectly by decreasing prey populations. For species that travel by sonar, noise produced by shipping, fishing, and coastal construction activities also has detrimental effects⁴⁴.

Despite these challenges, MPAs do have the potential to alleviate stress for mega fauna. A solution to one major issue, the migratory nature of mega fauna, is to protect sites where mega fauna species are the most vulnerable. In the case of shark species, i.e. elasmobranch species, MPAs have thus far been used to protect them during life stages where they are less mobile, i.e. when they remain in nursery areas^{45,46}. Site selection criteria have been formulated to increase the likelihood of having a site with high fidelity and thus increase the effectiveness of conservation efforts³⁷. However, studies have shown that the conservation yield of providing protection during the juvenile life stage of sharks is minimal⁴⁷. Protection of juveniles and younger age classes has a limited effect on population growth rates because most anthropogenic related mortalities or physical traumas occur in later life stages. Through the use of demographic models, it has been shown that the most important life-stage to protect would be juveniles nearing maturity⁴⁸⁻⁵⁰. However, during this life stage, their home range increases, diminishing the conservation value of protected areas that are stationary. Conservation benefits are lost once the protected organism migrates through waters that are subjected to sources of anthropogenic stress. An example of how MPAs are unable to protect migratory mega fauna is provided by the management of the hound shark (*Galeorhinus galeus*)⁵⁰. Despite the efforts to protect nurseries over a period of over 30 years, *G. galeus* populations have declined to the point of population collapse. Protecting the sharks within their nurseries was not sufficient to provide sustainable population growth.

Efforts in preserving marine mammals have used similar site selection methods in order maximize the conservation effectiveness of MPAs⁵¹. However their effectiveness remains the subject of much debate, in part because there are very few studies that provide empirical evidence of the effects of MPAs on marine mammals. Despite this lack of evidence, MPAs are considered to be an important tool in the preservation of marine mammals^{51,52}. The few studies that do provide evidence have shown limited influence of MPAs. The survival rate of the Hector's dolphin (*Cephalorhynchus hectori*) survival rate increased by 5.4% over a period of 20 years following the implementation of a reserve⁵³. The authors note, however, that their results are in line with other research showing that the MPAs are too small to provide sufficient and effective protection. Nonetheless, the use of spatial data allows for site selection in order to decrease mortality rates and injury. Such methods have been used in the Great Barrier Reef World Heritage Area for the protection of dugongs (*Dugong dugon*)⁵⁴. Through the rezoning of the Great Barrier Reef and location of activities that are detrimental to the environment to areas where dugongs are not present, the conservation effectiveness of the protected area has increased⁵⁵.

Other mega fauna species such as turtles have also been the subject of research in relation to MPAs. Spatial data has been used to show that turtle (*Chelonia mydas*) concentrations inside protected sites were at least four times higher compared to control sites⁴⁰. The underlying processes for higher turtle density within MPAs is, however, still open for discussion and might not be attributable to increases in survival, birth or growth rates. Population growth rates of green turtles are generally

low, and the sudden increase may be the result of higher rates of immigration¹⁸. Nel *et al.* (2013) conducted a survey of the population dynamics of the leather back

(*Dermochelys coriacea*) and the loggerhead (*Caretta caretta*) species. Different trends were visible for both species. Loggerhead species showed a dramatic increase over a time span of 45 years (Figure B.3), average nest count was almost 3 times higher at the end of the survey period. Leatherbacks showed a less pronounced trend, which was characterized by large annual variations throughout the survey.

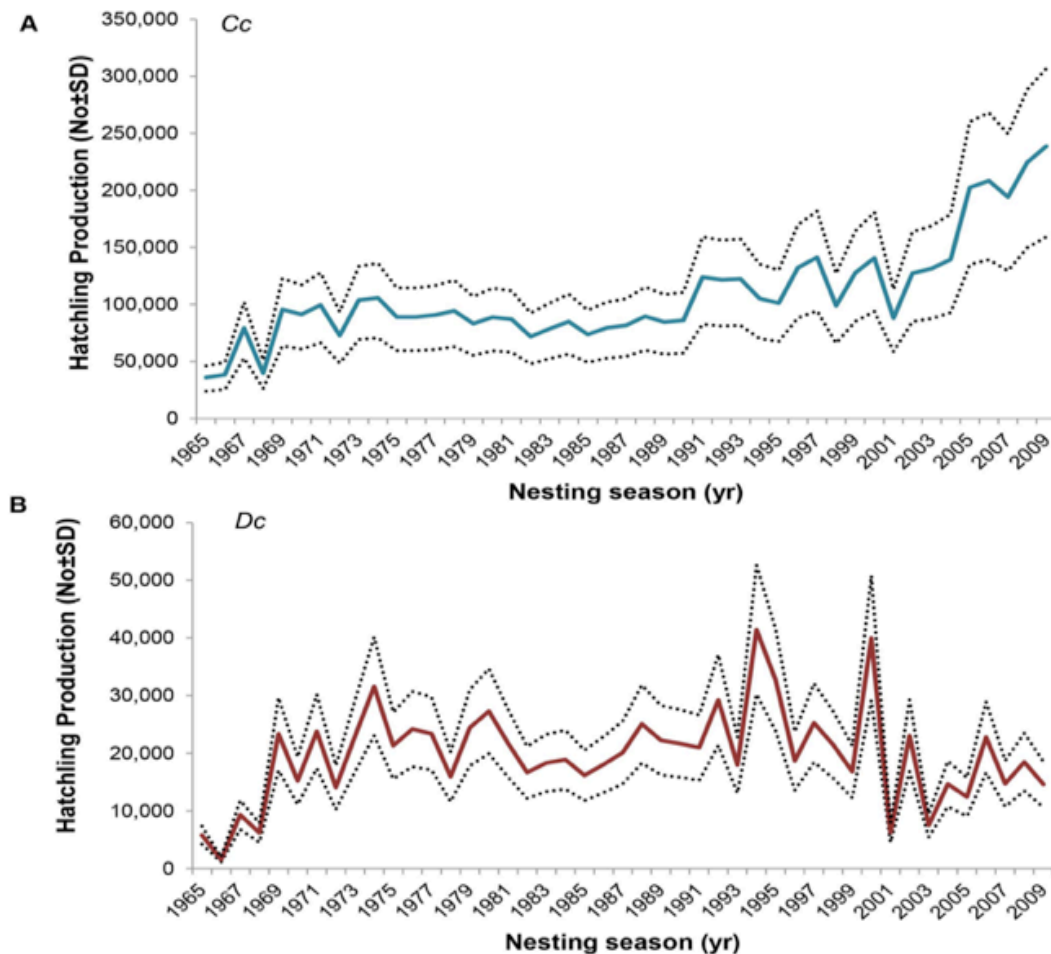


Figure B.3 Estimated hatchling production of loggerhead (Cc) and leatherback (Dc) turtle species over time (dotted line represents the SD). Implementation of MPAs occurred in 1963 and at another site in 1986. Nel *et al.* (2013).

Corals

As a result of the plethora of local and regional stressors present coral cover has been in decline since the 1990s⁵⁶⁻⁵⁹. Major contributing stressors are: overfishing, destructive fishing, siltation and pollution. By eliminating as many threats as possible, an MPA can stimulate the resilience of an ecosystem.

¹⁸ More is not always better. The hyper densities of the green sea turtle resulted in the near collapse of the seagrass ecosystem. MPAs thus also have potential negative effects. See Christianen (2014) for a fully detailed report.

Results of recent studies provide mixed evidence on the success of MPAs in this matter⁶⁰⁻⁶². A meta study conducted in the Philippines indicated that coral cover within MPAs increased annually with 3.2% since 1981⁶³ and was independent of MPA size, age, protection level (fully vs. partially protected). The study compared 56 MPAs to 57 non-protected areas. The results showed that on average MPAs had significantly higher coral cover compared to neighbouring fished areas: 36% compared to 30.2%. Another meta-study that conducted an analysis on 310 MPAs showed coral cover within MPAs to remain constant on average, whereas a steady decline was seen in non-protected areas⁶⁴. Caribbean corals fluctuated between 20-35% coral cover and Indo-Pacific corals between 10-20%. In the same region, non-protected areas showed respective average annual rates of decline of 0.3% and 0.4%^{64,65}. Studies conducted within the Great Barrier Reef have similar results, on average coral cover within MPAs is 24%, whereas fished areas 17%⁵⁵. It is interesting to note that outbreaks of the crown of thorn starfish, a natural predator of corals, was 3-7 times higher within fished reefs. Most meta-studies show that the annual change in coral cover within MPAs is either slightly negative or neutral, few studies in fact, show a positive trend (table 1). A study conducted by Olds, Pitt, and Maxwell (2014) showed that MPAs were creating more resilient ecosystems. After a serious flooding event in 2011 in Australia, coral reefs within MPAs recovered faster compared to coral reefs that were subjected to fishing activities. There were mass macroalgae blooms in the non-protected areas. The flooding created conditions less favourable for corals by increasing turbidity and nutrient concentrations (Figure B.4). Because fished areas had lower numbers of herbivorous fish compared to the MPA, macroalgae could grow unhindered⁶⁶.

Table B.3 The response of coral cover to MPAs. A comparison within and outside MPA establishment. All values are statistically significant

Author (Year)	N (MPAs)	Location	Coral Cover (%)		Annual Growth (%)	
			MPA	Non	MPA	Non
Magdaong	56	Philippines	36	30.2	3.2	0
Seleg	310	Indo Pacific	-	-	0	-0.27
Seleg	310	Caribbean	-	-	0	-0.43
McCook	12	Australia	23	17	0	0
Hargreaves	66	Global	-	-	-0.23	N.A.

Regarding the level of protection, a number of studies have shown that this has no effect on rates of coral decline^{67,67}. A study on coral cover in the Indian Ocean by Graham (2008) showed that corals within MPAs were just as susceptible to coral cover declines as unprotected areas.

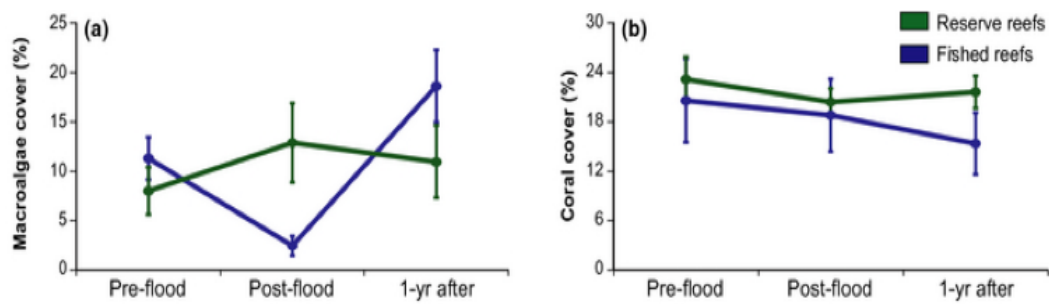


Figure B.4 Percentage cover of macroalgae (a) and coral (b) in fished reefs (blue) and reserve reefs (green). Graphs show the cover in pre- and post-flood conditions and 1 year after the flooding event. Source: Olds (2014)

Ecosystem resilience

To a greater or lesser degree all marine ecosystems are naturally dynamic and constantly undergoing changes, be it on short- or long-time scales⁶⁹. Under these natural conditions they have the capacity to maintain key processes that allow them to rebound from drastic changes. This concept is known as ecosystem resilience and it entails the capacity of a system to withstand stress and recover from persistent or frequent disturbances. However, when stress is too high and disturbances too frequent, the damage caused might be irreversible⁷⁰⁻⁷². In such a case, resilience is undermined and a complete phase shift can occur (Figure B.5). Figure B.4 provides a graphical representation of how stress and disturbance might interact in a simulated coral ecosystem. Depending on the frequency and relative abundance of stress, an ecosystem dominated by corals can completely change, and turn into an algae dominated system.

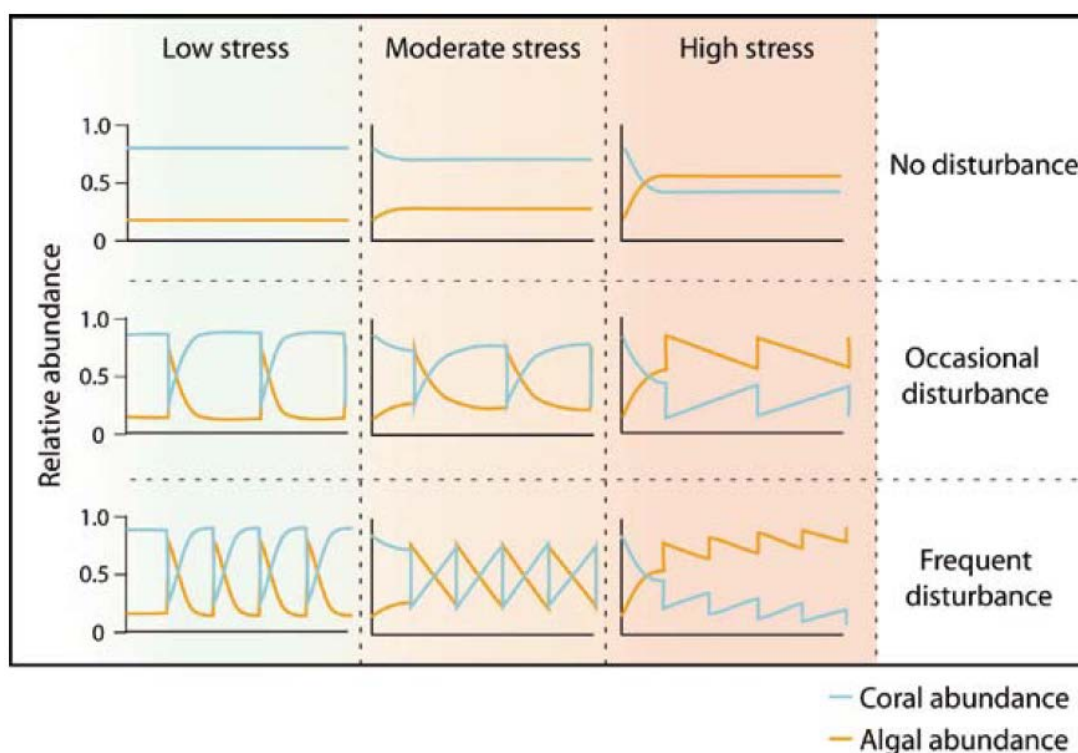


Figure B.5 A graphical representation of how stress and disturbance might interact in a simulated coral ecosystem. The graphs depict the change of a coral (blue lines) or algae (brown lines) dominated ecosystem in relation to stress and disturbances. Source: McCook, 2010.

Primary producers such as kelp, mangrove and seagrass species can also be influenced by the presence of MPAs. Such primary producers are important for ecosystem functioning, providing diverse habitat, nursery area, control sedimentation, and filter sediment runoff⁷³. Babcock and Kelly (1999) showed a complete urchin dominated area, returned to its original kelp forest state after 12 years of protection. Kelp biomass restoration resulted in an average increase of primary production by 58%. On a global scale seagrass and mangrove ecosystems are showing decline⁷⁴⁻⁷⁶. Some studies show that areas where there is no to little stress, seagrass and mangroves are in greater abundance (biomass/m²)⁷⁷. When sources of stress are removed, seagrass has shown to be able to recover⁷⁸. Multiple studies have shown that a reduction in water pollution can result in the recovery of seagrass beds⁷⁹⁻⁸¹, although in some cases over long time scales⁸².

It remains uncertain which characteristics of an MPA can increase the restorative capacity and resilience of an ecosystem. What is clear is that the presence of specific functional groups allows for a more rapid recovery of an ecosystem⁸³⁻⁸⁵. For example, in the case of algae dominated coral reefs, the return of herbivores allows the system to rebound⁸³. Restoration of primary producers is the result of alterations in abundance and compositions of herbivores. These alterations have effects that cascade throughout the ecosystem, indirectly benefiting primary producers such as reef building corals. Through the removal of local stress, functional groups such as predators and herbivores slowly re-inhabit their ecological niche and the system rebounds to its former pre-stress state. The study of Babcock and Shears (2010) showed just that. Urchin dominated kelp areas were seen to diminish once predator lobster species returned. This fits well within the premise that urchins come to

dominate habitat because of a lack of higher-level carnivorous organism¹⁹. It is interesting to note that no such recovery occurred in areas where fishing still took place. There are, however, a number of studies where MPA implementation has not proven to be effective within the context of increasing resilience or recovering destroyed habitat⁸⁷⁻⁹⁰.

Conclusion

There is substantial and growing evidence of the positive effects of MPAs on populations of individual organisms and ecosystems. Reducing anthropogenic pressures through establishment of MPAs has been shown to have positive impacts at every trophic level, from sea mammals and sharks to corals and kelps. In the case of species at higher trophic levels (i.e. sea turtles^{34,35}, mammals^{26,38,47}, and sharks²⁶), conditions become more favourable for their survival by directly decreasing hazardous activities that take place and indirectly because prey populations become more abundant. In some cases, fish abundance has been shown to be 14 times higher within an MPA. At lower trophic levels (i.e. corals^{63,64,69}, seagrass^{75,76}), resilience increases as the ecosystem becomes more diverse because functional groups such as herbivores and predators become more abundant. Specific MPA parameters such as age, size, protection level and location have rarely proven to be significant determinants of MPA effectiveness^{14-16,31}, although there are exceptions²⁰. It is more often the case that studies indicate no relationship with biological parameters. Furthermore, there is still a lack of evidence and understanding of the mechanisms through which MPAs affect biological parameters.

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Appendix C To what extent can changes in marine ecosystem service provision in response to MPA designation be quantified? A rapid literature review

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Abstract

It is commonly assumed that a variety of marine habitat types (e.g. seagrass meadows, kelp forests, coral reefs, mangroves) provide a range of marine ecosystem services (MES), and hence benefits to humankind. Despite there being substantial ecological evidence regarding the ecological value of these habitats, and a range of economic valuations capturing various elements of the benefits provided by these habitats, little work has been done, to date, to collate available information quantifying the provision of specific ecosystem services by specific habitats. This is a particularly relevant omission in the context of increasing the number of marine protected areas (MPAs) globally, as flows of ecosystem services provide an alternative to the pure conservation narrative that can sometimes surround MPA designations. This study presents the results of a systematic effort to survey existing information in the peer-reviewed literature on the quantification of service flows for several marine ecosystem services (e.g. seafood, climate regulation via carbon sequestration, coastal erosion prevention, disaster mitigation, and tourism/recreation) from marine habitats relevant to MPA designations. The results of this effort indicate that although reasonable (theoretical) progress has been made with respect to the argumentation surrounding the provision of MES, the kinds of empirical data necessary to estimate generalizable, quantitative relationships of MES provision are still either missing or scattered within the literature. Until these deficiencies are resolved, it will not be possible to assess the full MES “cascade” from ecosystem functions through to ecosystem services and values in the context of MPA designation.

Executive summary

The purpose of this study was to conduct a high level review of the literature in order to identify, wherever possible, evidence linking biophysical changes in the marine environment to changes in the provision of marine ecosystem services (MES) such that changes in MES provision could be estimated in the context of global MPA expansion scenarios. In the context of the larger project, this review sits in between a review conducted on the biophysical impacts of MPA designation and research on the economic valuation of changes in MES provision.

The searches were conducted systematically (see Appendix C1 for details), and yielded a wide variety of results in terms of the size of the existing literature pool. No relevant studies were found for a range of MES (such as Waste Treatment in the context of kelp stands), whereas more than 100 studies were returned for other MES (such as Lifecycle Maintenance in the context of mangrove forests). The documentation of quantitative relationships between biophysical changes and changes in MES provision was also highly variable across the different MES considered and the contexts considered. Table C1 provides a summary of the results in that it highlights for each MES considered whether or not a quantitative relationship between environmental /habitat change and MES supply was found. It also then highlights the particular sections of this report (and page numbers) pertaining to the discussion of each MES considered. Subject to the caveats and assumptions employed in the original studies used, the quantitative relationships that were found can be used to help inform the broad-scale economic valuation of global MPA expansion scenarios.

Table C1 High-level summary of those ecosystem services for which quantitative relationships were found

Ecosystem service	Habitat-specific relationship (if relevant)	Global quantitative relationship	Page (for relationship)	Accompanying report section
Seafood		Yes	123-124	3.1.4
Recreation & Tourism		No		3.2.1-3.2.7
	Seagrass beds	No		3.7.1
	Macroalgae Stands	No		3.7.2
	Mangrove Forests	No		3.7.3
Climate Regulation		No		3.7.4
	Seagrass beds	Yes	131	3.3.1
	Macroalgae Stands	Yes	132	3.3.2
	Mangrove Forests*	Yes	134	3.3.3
Erosion Prevention		No		3.3.4
	Seagrass beds	Unclear ¹		3.4.1
	Macroalgae Stands	No		3.4.2
	Mangrove Forests	No		3.4.3
Waste Treatment		No		3.4.4
	Seagrass beds	No		3.5.1
	Macroalgae Stands	No		3.5.2
	Mangrove Forests	No		3.5.3
	Coral Reefs	No		3.5.4

¹ None of the studies, at the abstract level, discussed presenting such a relationship. More time would be required to check, in detail, the contents of the cited studies to ensure no usable equation is present.

Table C2 High-level summary of those ecosystem services for which quantitative relationships were found (continued)

Ecosystem service	Habitat-specific relationship (if relevant)	Global quantitative relationship	Page (for relationship)	Accompanying report section
Lifecycle Maintenance	Seagrass beds	Yes	147	3.6.1
	Macroalgae Stands	No		3.6.2
	Mangrove Forests	Yes	149-Error! Bookmark not defined.	3.6.3
	Coral Reefs	Yes ²	150	3.6.4
Air Purification	Seagrass beds	No		3.8.1
	Macroalgae Stands	No		3.8.2
	Mangrove Forests	No		3.8.3
	Coral Reefs	No		3.8.4
Cultural Heritage & Identity	Seagrass beds	No		3.9.1
	Macroalgae Stands	No		3.9.2
	Mangrove Forests	No		3.9.3
	Coral Reefs	No		3.9.4
Raw Materials	Seagrass beds	No		3.10.1
	Macroalgae Stands	No		3.10.2
	Mangrove Forests	No		3.10.3
	Coral Reefs	No		3.10.4

* In the case of mangroves there is information not only on carbon sequestration within mangroves forests but also on the potential for carbon dioxide emissions from damaged or cleared mangrove forests

As this table shows, although there are some MES for which quantitative relationships of some description were found, there are many MES for which none were found. There are a variety of reasons why this is the case. One reason is data availability. Even in the case of MES that are well-defined and have an easily justified unit of measurement (such as tourism and recreation, measured by person-days), it is not always the case that the data has been collected and analysed over a large enough geographic scale to understand how biophysical changes lead to changes in MES provision beyond a few very site-specific case studies.³ Quantitative relationships linking biophysical changes to MES provision are also lacking because almost none of the existing literature reports on research that was intending to directly measure MES provision, and in many cases it is not possible to reinterpret what was measured/recorded in terms of MES supply (which frequently would be measured in different units).

In the context of needing to understand the impacts on MES provision of MPA designation, however, the primary factor warranting consideration is the design of MPA impact studies themselves. As explained in depth in sections C3.1 and C3.2, and reiterated throughout this appendix, the design of many of the existing studies that are focused on changes resulting from MPA designation are fundamentally inadequate

² This 'Yes' is conditional on the coral reefs being cold water coral reefs and a range of assumptions being acceptable in the context of the resulting scenario analysis.

³ As the Ocean Health Index project has also found, data on coastal tourist numbers is difficult to find. The Ocean Health Index has calculated index scores related to tourism and recreation for more than 100 countries around the world. However, a close reading of the method employed (OHI 2015) demonstrates that some large assumptions had to be made in order to estimate coastal recreational data from large, private domain tourist data sets that did not distinguish between forms or specific location of tourism.

to isolate the biophysical impacts of marine protected area designation. Because the biophysical impacts of MPA designation (in different ecological/geographical/management contexts) very often cannot be assessed clearly, it is also currently not often possible to measure, understand, or model changes in the supply of MES *resulting from these biophysical changes*. There are also virtually no studies focused on directly measuring/monitoring MES supply in marine contexts.

The ability to understand the relationships that exist in different contexts between MPA designation and MES provision would be greatly improved in the future by research that employs study designs capable of isolating and quantitatively measuring the biophysical impacts of MPAs, as well as by research that endeavours to directly measure MES provision through time (using biophysical units that are compatible with the MES definitions adopted). As studies of this nature increase in frequency, so too will the ability of researchers to understand the nature of MES provision, and by extension, also the economic valuation of MPA designation.

C1 Introduction

There is increasing interest in documenting (as well as understanding) the flows of marine⁴ ecosystem services (MES) from marine protected areas (MPAs) (e.g. Potts *et al.* 2014). This information is relevant not only because the concept of ecosystem services is becoming more relevant to management (e.g. the EU Marine Strategy Framework Directive - MSFD), but also because the narrative of ecosystem services is one that highlights the ways in which humankind benefits from healthy, functioning ecosystems. This narrative can provide a strong contrast to conservation narratives that may instead highlight the intrinsic value of ecosystems, or the uniqueness of certain ecosystems, rather than the anthropogenic benefits associated with those ecosystems. Despite this interest in understanding the relationship between MES and MPAs, however, efforts to quantify flows of marine ecosystem services in response to the implementation of marine management measures (including the designation of MPAs) are still fairly new in the literature.

Amongst the numerous contributing factors to this lack of clearly identified MES-MPA relations are each of the following:

1. Some uncertainty regarding the units to use when measuring MES provision
2. The availability of marine ecosystem data in *those* units or in units of good *proxy* measures for those units
3. Continuing scientific uncertainties regarding the linkages between different ecological components
4. Comparatively few studies conducted to date have expressly been focused on analysing marine environmental change through a quantitative MES lens

Consequently, in order to assess the extent to which definitive assessments can be made regarding ecosystem service flows from MPAs, it is necessary to first adopt a position on how MES can be measured and then contingent upon this decision, to analyse existing studies from wide range of disciplines and reinterpret the results of those studies through an MES lens.

The work presented here constitutes a rapid literature review contributes to this larger research requirement. Specifically, the purpose of this study was to conduct a high level review of the literature in order to identify wherever possible evidence linking biophysical changes in the marine environment to changes in the provision of MES such that changes in MES provision could be estimated in the context of global MPA expansion scenarios. In the context of the larger project, this review sits in between a review conducted on the biophysical impacts of MPA designation and research on the economic valuation of changes in MES provision.

Because the focus of the larger project is the analysis of global scenarios for the expansion of MPAs, it was necessary to consult literature from around the world. At the same time, however, due to the short term nature of the project, time exerted a significant constraint on the ability to conduct the review. The approach adopted (Section 2) endeavoured to balance between the competing requirements of the review: global coverage, multiple MES coverage, and rapid turnaround.

⁴ For the purposes of this report, the term 'marine' is used in place of the phrase 'coastal and marine'

C2 Methods

In order to conduct the review, it was necessary to have some understanding of how one might be able to measure, in biophysical terms, the provision of MES. In turn, this required adopting a particular ES definition and ES typology. This review utilized a recently published ES definition and typology with which the author was familiar and for which potential MES indicators had been suggested (Böhnke-Henrichs *et al.* 2013). The review that was then conducted focused on a subset of services. The services considered were: seafood, recreation, coastal erosion prevention, lifecycle maintenance, air purification, raw materials, recreation & tourism,⁵ and cultural heritage and identity. The list of MES considered partially reflects the known availability (or potential feasibility) of economic valuation for MES given existing data/studies. In other words, MES for which no economic value could defensibly be estimated using existing data (e.g. Inspiration for Culture, Art, & Design) were excluded from consideration due to the objectives of the study. The list of MES considered also partially anticipated importance of MES.

Information on MES provision for these ESs was sought using two different approaches. The first involved searching directly for studies analysing the impacts of MPAs on MES provision. This approach was adopted with respect to a variety of services (see section C, Table C2), but the most useful results related to seafood and tourism. Upon seeing the results, it was decided that the results for the other services were most usefully considered in the context of the habitat-specific literature (described below), and so were combined with those studies after the first round of filtering for relevance.

The second approach focused on MES provision from specific habitats that are often the focus of MPA designations (i.e. seagrass beds, macroalgae stands, mangrove forests, and coral reefs). The logic behind this choice was as follows: if there is evidence that MPA designation results in the ecological recovery (either via improvement in quality or extent) of one of these habitat types, and evidence can be found that those habitat types are known to provide certain MES, then changes in MES provision from the designation of MPAs could be inferred, at least broadly, in the context of the scenario analysis featured in the larger project. The MES targeted through this second approach were as follows: climate regulation, erosion prevention, waste treatment, lifecycle maintenance, air purification, recreation and tourism, raw materials (with respect to seagrass), and cultural heritage and identity.

It is worth noting that a third approach involving trying to identify changes in MES provision related to changes in marine mega fauna was tested given that the preceding section of this report engages with the literature related to MPAs and mega fauna. The tests in the literature were conducted with respect to sea turtles because some evidence was found that at least sea turtle concentrations increased within MPAs. The searches conducted are included in Appendix C1. However, few articles were returned that passed the first stage of filtering, and upon subsequent investigations, none of the articles returned were deemed to be useful in the context of this report. Because of this and given the time constraints, this line of inquiry was not taken further.

This review was conducted *systematically*, but does not by any means constitute a Systematic Review (as defined by the Center for Environmental Evidence)⁶ and so makes no claims to being exhaustive even within the peer reviewed literature. Instead,

⁵ Hereafter referred to as 'tourism'

⁶ <http://www.environmentalevidence.org/information-for-authors>

this review constitutes a rapid and high-level overview of the literature relevant to understanding how MES provision changes with changes in marine environments. All literature searches were conducted in ISI Web of Knowledge (WOK)⁷ using search strings intended to identify literature that would document some aspect of the relationships between biophysical parameters in marine systems and MES provision (Appendix C1). All the searches were conducted with reference to the “topic” (i.e. title, abstract, and key words) of the studies.

These search terms were intended to be simple and to have a tendency towards broad inclusivity. This was thought to be important given the lack of time to formally test and refine the search strings, as one would within a formal Systematic Review. An effort was also made to conduct the same searches across each of the habitat types considered, though in some cases it was necessary to modify the searches slightly and some adaptation of search strings occurred in certain circumstances. Because of time constraints, the date range was also restricted to studies published in the 1994-2014 timeframe. In the final searches conducted (in reference to culture), it was necessary to additionally restrict the results to those abstracts classified as “social science, arts, and humanities” abstracts in order to exclude all the results related to microbiology and the culturing of bacteria in labs. Finally, it is important to note that because of time constraints, these searches were not repeated in other databases and grey literature was not sought.

Studies returned through these search efforts were firstly vetted based on title and abstract contents. Initially, studies were excluded if they were spurious results (i.e. from completely unrelated fields), if they did not actually relate to either MPAs or marine ecosystems, or if they appeared to be completely conceptual/theoretical in nature. Studies were also excluded that appeared to belong exclusively to the purview of the previous and subsequent parts of the large project (i.e. the biophysical impacts of MPA designation and the economic valuation of MES), though in some instances this was not clear from the first inspection of the study. The searches returned literature that fit, by and large, within the anticipated themes. However, in some instances a search targeting one MES returned abstracts that were actually more relevant to another MES. When this happened, those studies were passed through the first round of filtering and saved for later consideration. Because of time constraints, these searches were conducted sequentially (see Appendix C1) and not independently. The ‘Marked List’ and EndNote Web features of WOK were utilized to identify those studies that had already passed through the first round of filtering, thereby largely enabling the researcher to avoid the consideration of duplicates across the overlapping searches. This decreased the amount of time required to search through the literature, but is another deviation of from the method one would need to employ to conduct a Systematic Review.

Due to the rapid nature of this review, and the very diverse set of literature returned, it was not possible to develop quality-related filtering devices to further narrow the field of literature under consideration, and the resulting collection of studies could not be read in full or mined deeply for data. Consequently, the following approach was adopted: all the abstracts retained after this first round of filtering were then grouped according to the MES to which they were most relevant, and were then re-assessed in a second round of filtering in order to 1) identify those studies for which the full text needed to be consulted and 2) extract key results, quantitative relationships, and key conclusions.

⁷ <http://wok.mimas.ac.uk/>

The following strategy was adopted to deal with the volume and diversity of literature produced through this process:

- Abstracts were considered in order of most recently published to least recently published
- Global (or regional) reviews, meta-analyses, and modelling studies that appeared promising in terms of the potential for them to include quantitative relationships were read in full, but were not mined for further references⁸
- When an abstract of a study (that was not a global review or meta-analysis) presented results directly relevant to the quantification of service flows, this information was taken at face value⁹
- When an abstract appeared to indicate that the full paper presented results directly relevant to the quantification of service flows, but these results were not themselves present within the abstract, the full paper was considered
- When a closer inspection of an abstract revealed a focus that was not relevant to the issue of the quantification of service flows, potentially useful contextual information was noted, and the sources were not dealt with further
- When an abstract repeated a theme or idea that had already been documented by a variety of abstracts considered (e.g. that SCUBA divers can damage coral reefs through contact and breakage), the abstract was not considered further because it was not deemed to add anything new to the evidence already collected

It is worth noting that the time constraints on this part of the project were such that it was not possible to conduct original meta-analyses on collections of single-site case studies. Consequently, priority consideration was given to existing global reviews and meta-analyses, whereas single-site case study data was used to help inform the broader analytical picture and functioned as tangible, illustrative examples.

Overall, despite some inherent weaknesses due to the inability to be exhaustive, to apply critical quality metrics to the studies considered, and due to the inability to consider the full text for all the studies included after the first round of filtering, the approach adopted did enable the researcher to survey of an extremely wide pool of literature covering a variety of MES. As such, this review should be considered as constituting a baseline the subsequent analyses that can employ more rigorous and thorough approaches.

⁸ The only partial exception to this was as follows: A number of modelling studies considered early in the process of conducting this review identified a number of empirical studies as presenting convincing empirical evidence related to spillovers (i.e. Abesamis et al. 2006; Abesamis and Russ 2005; Alcala et al. 2005; Davidson 2001; Galal et al. 2002; Goni et al. 2006; Grafton et al. 2005; Kelly et al. 2002; McClanahan and Mangi 2000; Murawski et al. 2005; Roberts et al. 2001; Russ 2002; Russ and Alcala 1996). Some of these references appeared in the searches conducted and some did not, but they were all considered.

⁹ The lack of critical assessment criteria is more relevant to the creation of a Systematic Map than a Systematic Review. see for example Randall and James (2012).

C3 Results

MPAs and Ecosystem Services

The first set of results presented here focuses on the outcomes of the literature searches that targeted the MPA literature specifically (as opposed to habitat-focused literature). The search strings used that returned results, as extracted from Table C1.1, are shown below (Table C2). As Table C2 shows, searches were conducted for a broader range of services than just seafood and tourism and recreation. Unlike the results related to fisheries yields and tourism, however, it turned out that the studies returned by these searches were most usefully combined with and considered alongside the literature returned from the habitat-specific searches. Consequently, this section of the report focuses only on the relationship between MPAs and fisheries yields, on the one hand, and the relationship between MPAs and tourism, on the other.

Table C3 MPA-focused search strings used (see Table C1.1 for full details)

Intended MES link	Search string used
Seafood	Catch* AND Marine Protected Area CPUE and Marine Protected Area (Marine Protected Area OR Marine Reserve) AND CPUE (Marine Protected Area OR Marine Reserve) AND (Spill over and spillover)
Ornamental Resources	Sea shell AND Marine Protected Area
Tourism & Recreation	Recreational Fishing AND CPUE AND Marine Protected Areas Marine Protected Area AND touris* (Marine Protected Area OR Marine Reserve) AND (tour* OR recreation*)
Lifecycle Maintenance	(Marine Protected Area OR Marine Reserve) AND Nursery
Climate Regulation	(Marine Protected Area OR Marine Reserve) AND (carbon sequestration OR carbon export)
Waste Treatment	(Marine Protected Area OR Marine Reserve) AND Waste
Coastal Erosion Prevention	(Marine Protected Area OR Marine Reserve) AND (Erosion OR wave propagation OR wave attenuation OR coastal protection)

C3.1 MPAs and fisheries (i.e. seafood provision)

It is often anticipated that MPA designation will help to both secure and increase fishing yields (i.e. the provision of seafood) by shielding a portion of the population from the threat of extraction (Higgins *et al.* 2008; Russ *et al.* 2004; Tupper and Rudd 2002). The logic behind this idea is that as populations within any given MPA recover in the absence of anthropogenic extractive pressures, the number, age, and size of individuals within the MPA will increase, as will the export of larvae into the fished areas (Russ 2002). In turn, this will lead to density-dependent spillover into the unprotected waters around the MPA and an increasing number of juveniles within the waters around the MPA. Both the spillover and the larval recruitment can, in theory, lead to increased number of fish caught, increased average size of fish caught (and therefore increased financial value), increased overall catch by weight, and increased

catch per unit effort (CPUE) (Figure C1). In other words, it is considered to be at least theoretically plausible that MPA designations will create a win-win situation favouring both conservation and fisheries.

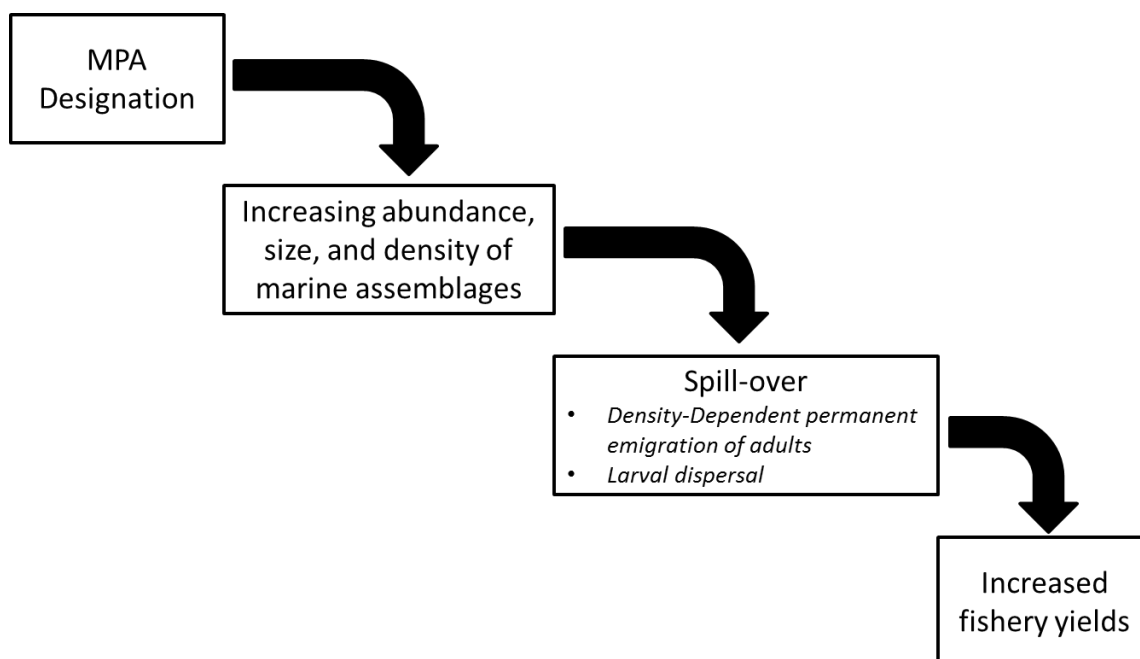


Figure C1 Anticipated cascade from MPA designation to improvements in fishery yields

The results of the literature review conducted, however, indicate that it is quite difficult to adequately achieve (and document) the cascade of impacts shown in Figure C1.¹⁰ Specifically, the literature returned by the searches conducted shows the following:

- The spillover effect that is a pre-condition for the stated fisheries benefits does not always occur following MPA designation, even when the populations within an MPA do change as anticipated (i.e. increasing in numbers, size, and age)
- When the required spillover does occur, the result is not always increased fish catch when compared to pre-MPA catch levels (and depending on the sustainability of the baseline yields, such an outcome may/may not signal something positive about the MPA and/or the fishery in question)
- Catch per unit effort (CPUE) ceases, in some circumstances, to be correlated with abundance inside the MPA after MPA designation, meaning that in these circumstances data on abundance data cannot be used to infer anything about the CPUE, or by extension actual fish catch
- In the absence of an actual measure of total effort, changes in CPUE cannot be used to unequivocally demonstrate increased yields following MPA designation
- There are various characteristics of MPAs and the contexts in which they have been designated that serve to confuse the detection of a relationship between MPA designation and fisheries yields if they are not documented and controlled for

¹⁰ Please also see Higgins et al. (2008)

- There are various features of existing MPA impact studies that frequently make it impossible not only to verify increased yields, but also to generalize a relationship between MPA characteristics and changes in yield

Each of these points is elaborated on below, the primary consequence being that it is almost always impossible to quantify MPA-yield relationships from existing empirical data as it has been reported in the peer-reviewed literature, and it does not seem to be possible to generalize an MPA-yield relationship from existing empirical studies. What this means is that the relationship between MPA designation and fisheries yields is currently best explored through modelling studies.¹¹ Given the literature found, the use of equations from modelling studies is what is recommended in this report, and to this end a couple of different alternatives are presented in section C3.1.4.

C3.1.1 Evidence related to the occurrence of the spillover effect

Spillover is a term typically used to refer to permanent adult emigration (i.e. density-dependent spillover), though it does essentially also include larval export/dispersal from protected areas into unprotected waters. Although larval dispersal may have a larger impact than density-dependent spillover on recruitment outside the MPA, it is very difficult to measure or detect (Francini-Filho and Moura 2008), so the primary focus to date has been the detection of density-dependent spillover.

There are some studies that are frequently identified in the literature as having presented relatively strong evidence that density-dependent spillover did occur following the designation of MPAs around the world (i.e. Goni *et al.* 2006; Grafton *et al.* 2005; McClanahan and Mangi 2000; Murawski *et al.* 2005). The broader MPA literature suggests, however, that spillover is the by-product of the interaction of a wide variety of context-specific factors beyond MPA designation (Box C1), and quantifying the spillover effect requires that a wide variety of social and ecological features be monitored.

*Box C1 Features that affect the likelihood of spillover occurring following MPA designation (Blyth-Skyrme *et al.* 2006; Brochier *et al.* 2013; Freeman *et al.* 2009; Ludford *et al.* 2012; Mesnildrey *et al.* 2013; Oresland and Ulmestrand 2013; Perez-Ruzafa *et al.* 2008; Pillans *et al.* 2005; Tupper and Rudd 2002)*

- Pre- and post-exploitation levels
- MPA size, shape, and age
- MPA management context
- Time since designation
- Species life history traits
- Species ecological traits, including home range, mobility, and maturation rates
- Whether MPAs prompt changes to the residency behaviour of species
- Habitat traits including connectivity, health, and circulation patterns

With respect to the monitoring of relevant ecological features, one study goes so far as to argue that it is essentially impossible to quantify the spillover effect unless the “full complexity of fish life histories” consideration (Brochier *et al.* 2013). Other studies contend that spillover effects cannot be quantified without considering the difference between spillover (permanent emigration), immigration, and “leakage” (the day-to-day

¹¹ As highlighted in studies (such as Higgins *et al.* 2008), however, exploring the MPA-fisheries relationship through modelling studies is also difficult, and it is certainly a less than ideal solution to extrapolate from a single model constructed for a single purpose to global scenario analysis.

crossing of MPA boundaries in both directions as a consequence of the MPA being smaller than a species home range) (Goni *et al.* 2006; Perez-Ruzafa *et al.* 2008). Given this, it is not surprising that few studies (i.e. some modelling studies and fewer empirical studies) have actually *quantified* spillover effects, let alone derived validated quantitative relationships to predict the magnitude of spillover effects.

Most studies that discuss spillover effects, including the frequently cited studies mentioned above, have instead *inferred the existence* of the spillover effect from a limited sub-set of (frequently exclusively ecological) indicators. Inference of this sort is considered by some research to be equivocal (i.e. Follesa *et al.* 2008) rather than conclusive, and some argue that as of 2002 no study had unequivocally empirically *quantified* a spillover effect from MPAs (Russ 2002; Russ *et al.* 2004). One notable exception to this trend is Goni *et al.* (2010). This study quantified both spillover and then analysed the impact on fishery yields, and found that between years 8 and 17 of protection the spillover of a lobster (*Palinurus elephas*) from the Columbretes Islands Marine Reserve equated to an annual, mean benefit to the fishery of 10% by weight.¹² There are also modelling studies that have quantified spillover in a particular context (e.g. Brochier *et al.* 2013).

Other studies have tried to identify relationships related to proxies for spillover. One of the commonly measured indicators that is taken as a proxy for density-dependent spillover (and sometimes even yield) is catch per unit effort (CPUE). Although it is common to measure CPUE both inside and outside MPAs, it is rare that researchers try to identify or generalize a relationship between any of the particular features of an MPA and CPUE. One exception to this (i.e. Stelzenmuller *et al.* 2009), estimated a relationship between the Shannon-Wiener diversity index H_f (Eq. C1, based on pooled data from 42 species spanning all functional groups) and CPUE. The relationship was estimated twice: once in the context of an area less than 2 km from the edge of the MPA (Eq. C2, adjusted- $R^2=0.21$) and once in the context of the area greater than 2 km from the edge of the MPA (Eq. C3 adjusted- $R^2=0.25$).

$$H_f = \sum_{i=1}^R p_i \ln(p_i) \quad (\text{Eq. C1})$$

$$H_f = 2.35 + (0.24 * CPUE) \quad (\text{Eq. C2})$$

$$H_f = 2.09 + (0.31 * CPUE) \quad (\text{Eq. C3})$$

These relationships were, however, derived for a single MPA (around Medes Island in the north-western Mediterranean Sea), with respect to the selection of species caught on a single type of benthos (soft bottom sediments in <30 m of water), using a single gear type (trammel gear), and do not include any actual data conclusively connecting CPUE changes to changes in overall yield. This inherently limits not only the extent to which these estimated relationships can be applied more generally than the original study site, but also the extent to which these relationships can be used to infer anything about the impact of MPA designation on yields (at least without other assumptions regarding the nature of the relationship between CPUE and overall yield).

¹² It is worth noting that this study claimed to have been the first to actually “quantify the number and biomass of individuals annually spilling over from an MPA and their contribution to the local fishery catches” (Goni *et al.* 2010). The recentness with which that statement was made (i.e. based on data collection that finished in 2007 for an article published in 2010) highlights that the focus on quantifying spillover is relatively new in comparison to documenting the types of biological responses to MPA designation discussed in the previous chapter of this study.

Other studies have endeavoured to estimate relationships between pre-MPA CPUE and post-MPA CPUE and time in order to infer both that changes in CPUE pertain to changes in spillover, and that changes in spillover relate to changes in yield. Follesa *et al.* (2008) is a good example of this type of study in that it shows there is some ability to estimate how CPUE (defined in this study as kg of species caught per 50m of trammel net per boat) changes with time with reference to a baseline figure (Figure C2).

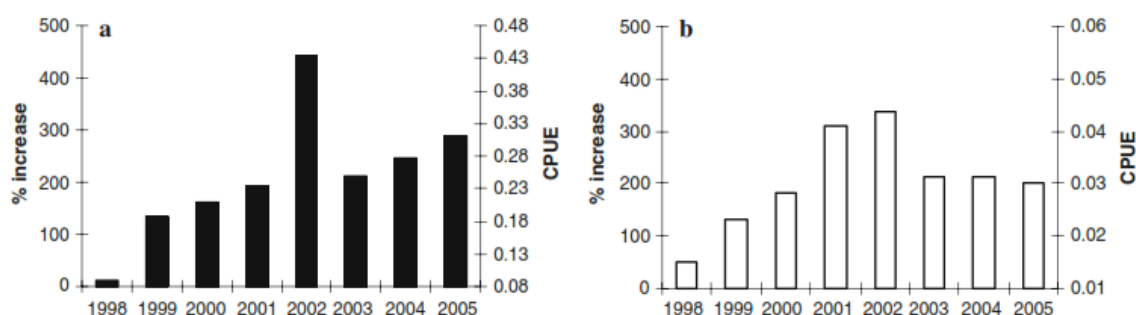


Figure C2 This figure is originally figure 2 from Follesa *et al.* (2008), where it has the caption “Rate of increase (%) of *Palinurus elephases* abundance inside (a) the reserve area and in the surrounding zone (b)

Even in this context, however, the pattern of changes in CPUE with time can only be interpreted as indicating that spillover effects have occurred if one also assumes that all other factors that might affect CPUE remain constant across the time frame considered.

Finally, it is worth considering the interactive effect of some of the other features in Box 1 in terms of the relationship between MPA designation and the occurrence of spillover. For example, there is evidence that the interaction between MPA size and species’ ecological traits has a non-trivial effect on the potential for spillover to occur. For example, spillover tends not to occur when species are fairly sedentary and/or have a small home range relative to the size of the MPA in question. Evidence of this has been found by studies focusing on the protection of various lobster and crab species and the picture of emigration from MPAs (or more appropriately the lack thereof) to unprotected waters painted by tag-release studies (e.g. Freeman *et al.* 2009; Moland *et al.* 2013; Pillans *et al.* 2005).

Spillover stemming from larval dispersal in relatively sedentary species may also be undermined by an interaction between the size of an MPA and larval dispersal distances, and potentially also the extent to which the habitats within an MPA are connected (or not) to sites that are conducive to larval settling (Little *et al.* 2007; Ludford *et al.* 2012). Modelling studies on the interactive effects of MPA designation and larval-dispersal support this contention. McGilliard and Hilborn (2008), for example, used a spatially-explicit model to investigate the impact that the distance of larval dispersal has on abundance, catch and CPUE (based on exploitation rate, abundance, number of boats and effort time) in the context of a fishery managed using a no-take MPA and effort control outside the MPA in the form total allowable catch (TAC) limits. Their results showed significant declines in CPUE relative to CPUE at

maximum sustainable yield (MSY).¹³ Although these declines were more severe when larval dispersal distances were small as compared to the size of the MPA, the model outputs still showed declines in CPUE for species with long dispersal distances. Critically, however, in this study *abundance* of the target species did not decline within the MPA after MPA implementation. Instead, abundance ceased to correlate with CPUE after the designation of the MPA (McGilliard and Hilborn 2008). This highlights that there may be at least some instances when it would be inappropriate to try and anticipate the later by measuring the former (at least in isolation from other variables).

Conversely, when MPAs are so small relative to the home range of the target species, that individuals within that species will find it impossible to remain within the MPA, MPAs will also not generate an internal increase in abundance or density-dependent spillover into the surrounding waters (Tupper and Rudd 2002). It is also worth noting that even if MPAs that were sufficiently large were designated for species with large home ranges, density-dependent spillover still may not occur as population size would have to increase significantly before density-dependence would force emigration from the MPA. This would be the likely case, for example, if MPAs were used as tool for managing and preserving the Green Jobfish (*Aprio virescens*) (Meyer *et al.* 2007).

Overall, therefore, although there is some evidence that spillover can occur following the designation of an MPA, this evidence is equivocal. The existence of the spillover effect depends on a variety of context-specific features, and its existence cannot necessarily be inferred from CPUE data in the absence of other corroborating evidence. Furthermore, no generalized empirical relationships were found that could estimate the magnitude of the spillover effect across contexts. This means that at least empirically, important parts of the cascade between MPA designation, abundance, spillover, and yield (i.e. seafood provision) remain insufficiently specified in the existing literature to be applied in the context of analysing the MES impacts of global MPA expansion.

C3.1.2 Spillover and fisheries yield

Where the preceding section discussed a lack of clear evidence documenting a connection between MPA designation and the *occurrence* of the spillover effect, this section highlights that there is also a lack of consensus in the available evidence regarding the relationship between spillover and measurable changes in yield (Stelzenmuller *et al.* 2009), a notion supported by the results of a recently published qualitative meta-analysis on MPA-fisheries linkages (Mesnildrey *et al.* 2013).

Some studies, for example, present evidence that the designation of an MPA has a positive impact on local fisheries through the spillover effect. For example, ten years after the designation of the Guokamma MPA in South Africa, the CPUE associated with the roman fishery (*Chrysoblephus laticeps*) was twice that documented prior to the designation of the MPA. There was no evidence of a systematic drop in total catch or in fishermen needing to travel increasing distances to achieve this increased CPUE (Kerwath *et al.* 2013), implying a real benefit to fisheries. Data from the area surrounding the Mnzazi Bay Marine Park in Tanzania indicates that between 2006 and 2010 (a time period that overlapped with the functioning of the MPA) the area supported an increased number of fishers, increased catch, and increased CPUE

¹³ In the case of species with short larval dispersion, CPUE declined to just 9% of CPUE at MSY, and for species with long larval dispersal distances, CPUE declined to ~50% of CPUE at MSY. For species with short larval dispersal distances, catch declined “substantially,” and for species with long larval dispersal distances “catch declined to values below maximum sustainable yield (MSY), but stabilized” (McGilliard and Hilborn 2008).

despite decreased time spent fishing (Machumu and Yakupitiyage 2013). Similarly, once an MPA equivalent to 15% of the area of the fished waters was created with respect to spear fishing in Bonifacio Straight Natural Reserve, Rocklin *et al.* (2011) found that CPUE increased by 60% seven years later, though the benefits were not uniform across all species. Evidence of recreational fisheries benefiting from MPA designation comes from Florida, where recreational catch of trophy fish species in two MPAs were significantly greater than the recreational catch from non-MPA areas (Bohnsack 2011). These studies, and others like them, do not present sufficient information to model the effects of MPAs on fisheries, and do not quantify spillover, but do provide some evidence that MPAs can have a positive effect on fisheries (and therefore that spillover is occurring at a sufficient level to supplement fisheries).

However, the relationship between spillover and yields is not, in many instances, particularly straight forward to assess. One reason for this is that it is unclear over what scale spillover actually operates. Some research suggests, for example, that the spillover effect operates only over a very limited spatial scale, and by extension its impact on fisheries can only be finite (Francini-Filho and Moura 2008). Other research suggests, in contrast, that while density dependent spillover can operate over a wide variety of spatial scales ranging from a couple of meters to several kilometres, larval dispersal can actually occur over significantly longer distances (Russ *et al.* 2004). This *variability* in the spatial scale of spillover (which is dependent on species-specific characteristics) has a number of effects in relevant to the issue of trying to understand the relationship between spillover and fisheries yields. Firstly, it might mean that there ceases to be the expected coincidence between the location of fishing and spillover, particularly if fishermen cluster their effort as a consequence of their *expectations* regarding spillover (known as “fishing the line”) (Kellner *et al.* 2007). Secondly, there is at least the potential for there to be a mismatch between the scale over which spillover is occurring and the effective boundaries of the fishery, something that would affect the ability of an MPA to compensate, through spillover, for a loss in fishing area.¹⁴ Finally, it means that spillover may not be detected if there is a mismatch between the spatial scale over which spillover operates and the spatial scale of an empirical study attempting to document the occurrence of the spillover effect and its effects on fishing.

Other research highlights that in the case of species with home ranges that are large relative to the size of the MPA designated, that any changes (in population size, spillover, or catch) due to the MPA are difficult to detect (Kellner *et al.* 2007), and that even in less mobile species the spatial heterogeneity of responses to MPA implementation across similar ecological systems highlights the complexity of the relationship between MPA designation, spillover and catch (Moland *et al.* 2013). By extension, this may also point to some non-trivial limits in the transfer of ecological production functions from one case study to more general analysis of MPA impacts. Overall, therefore, although there is some empirical evidence that spillover both occurs following MPA designation and that this spillover increases yield, this outcome is not a foregone conclusion. The literature contains case studies documenting highly variable

¹⁴ The issue of spillover compensating for decreased fishing grounds is an important one. Some studies (e.g. McClanahan and Mangi 2000) have documented instances where MPAs remove so much of the fishing grounds from use that fish catches decline severely relative to their starting levels even when spillover is present. Some research contends this may be due to the size of the MPA relative to the characteristics of the target species, and that small MPAs (i.e. $\leq 6 \text{ km}^2$) designated for species with limited mobility rates may lead to increases in commercial catch because the reduction in fishing area associated with the designation of small MPAs is minor relative to the potential for emigration (Follesa *et al.* 2008).

responses and insufficient information to quantify any kind of generalized relationship between spillover and changes in yield.

C3.1.3 Barriers to understanding the impact of MPAs on yields

In addition to there being some uncertainty as to when spillover occurs and under what conditions spillover can increase fishery yields, there are a range of other factors that serve as barriers to understanding (and therefore being able to effectively quantify) the relationship between MPA designation and yield. These factors (listed below) effectively relate to a number of features associated with the design of MPA impact studies. Each of the following are discussed briefly in this section:

- ➔ Explicit Study Design
- ➔ Duration of MPA studies & insufficient data on fish recovery
- ➔ A lack of baseline data and counterfactual analysis
- ➔ Confounding effects are often not been controlled for in study design
- ➔ Insufficient data on fish
- ➔ Insufficient data on fishing activity in the context of CPUE data
- ➔ The multi-faceted nature of MPA impacts

Explicit study design

Firstly, it is difficult to achieve a study design in the context of MPAs that can truly facilitate impact assessment (i.e. a before-after-control-impact (BACI) design), and this kind of study is largely absent in the literature focused on the fisheries impacts of MPA designation (Follesa *et al.* 2008; Goni *et al.* 2006), though there are some quite recent examples too (e.g. Clarke *et al.* 2014). The lack of BACI studies within the MPA literature means that effect size in terms of fisheries (or other potential MPA impacts) is difficult, if not impossible, in many instances to empirically measure. There appears to be good potential, however, for the use of quasi-experimental designs, however, using statistical matching procedures to identify pseudo-control sites to aid in MPA impact assessment (e.g. Ahmadi *et al.* 2014). Although this kind of study is also not very common within the existing literature considering the impact of MPAs on fisheries (Gurney and Pressey 2014), presentations delivered at the International Marine Conservation Congress (IMCC) in August of 2014 indicate there is, perhaps, increasing interest in drawing on the techniques developed within medical impact assessment and in utilizing pseudo-control sites to improve the quality of MPA impact research in the future.¹⁵

Duration of MPA Studies & Insufficient Data on Fish Recovery

Additionally, to date many studies seeking to analyse the impacts of MPAs have access to fairly short term data sets (Goni *et al.* 2006). A fairly extreme example of this trend is (Parnell *et al.* 2007), which considered fishing effort and catches over a single season. Given that there is some evidence that at least reef fish require a medium-to-long term recovery period (McClanahan *et al.* 2007), and that there is some evidence that recovery data some species is missing (Follesa *et al.* 2008), the implication that follows regarding short-duration MPA studies is that they are likely to be capable of shedding light on only a limited part of what is a larger impact picture, and therefore cannot be used to understand or quantify generalizable relationships between MPAs and fisheries yields.

¹⁵ E.g. Ahmadi *et al.* (2014); Gurney and Pressey (2014)

Lack of Baseline Data & Counterfactual Scenario Specification

A range of other studies highlight that a lack of baseline data also constitutes a nontrivial barrier to the assessment MPA-fishery impacts (e.g. Follesa *et al.* 2008; Goni *et al.* 2006). The lack of baseline data prevents researchers from conducting before-after-control-impact (BACI) studies (Pillans *et al.* 2005). In turn, this effectively makes the interpretation data that may otherwise appear to be highly demonstrative of positive MPA impacts on fisheries difficult. Russ *et al.* (2003) serves as a good illustration of this point. This study found that biomass within a band of space 200m outside the MPA had increased by a factor of 40 within 20 years of MPA designation and that 62.5% of fish catches occurred within this band. Although this appears to strongly support the idea not just that MPA implementation led to spillover, but also that spillover has increased fisheries yield, the results in terms of the impact on yield are still equivocal because the research did not have baseline data on the spatial distribution of fishing effort or fish catches. Essentially, although the results appear to be highly indicative of an increase in yields resulting from MPA implementation, this cannot actually be unequivocally confirmed without documentation of spatially-explicit reference data. Similarly, other research argues that even when considering gradients in fish abundance or size, unless the same gradient was assessed prior to MPA implementation (and habitat quality assessed), the gradients measured post-MPA implementation cannot really be interpreted clearly (Francini-Filho and Moura 2008).

Additionally, it is often the case that baseline *trends* (i.e. counter-factual scenarios) are not featured in the analysis of MPA effects (in either empirical or modelling studies). This is particularly relevant to the issue of assessing the impacts of MPAs on fisheries yields in contexts where MPAs are being pursued as a reactive measure in the response to increasing pressure on declining marine environments and or unsustainable levels of human activity. The relevant question in such cases when assessing the true impact of MPAs on fisheries yields is not how the yields post-MPA implementation compare to the yields of the relevant fishery (or fisheries) immediately prior to implementation, but how the yield post-MPA implementation compares to what would most likely have been the yield had the MPA never been implemented.¹⁶

Consider, for example, a purely hypothetical fishery that is not overfished, but for which an MPA is implemented. If the species is not in a state that requires recovery, yields would not be expected to increase with the MPA relative to the counter-factual scenario without the MPA (Gerber *et al.* 2003). Alternatively, consider a purely hypothetical fishery that is currently unsustainably fished and for which an MPA is being considered as a potential management tool. With the implementation of the MPA, it is possible that yields will decline and may never reach the same level as what was being caught prior to designation. This could be true, for example, in a fishery dominated by illegal fishing (e.g. Ainsworth *et al.* 2012) that is subsequently prevented by the enforcement of MPA management. Relative to the baseline yield, in this case it would appear that the MPA has had a detrimental effect on fisheries yields.

However, *if* the full counter-factual scenario is continued fishing at unsustainable levels followed by a severe decline in yield and/or stock collapse (dashed line, Figure C3), whereas the MPA leads to a reliable yield (solid line, Figure C3), then the MPA actually has had a positive impact on fisheries yields even though yields are lower with the MPA than immediately prior to MPA designation. There may be less extreme examples where this result could be true as well (Figure C4). Results such as these will likely not be detectable MPA studies that are not sufficiently long-term to evaluate

¹⁶ The importance of counterfactual scenarios was also raised at the recent 2014 IMCC (e.g. Kininmonth *et al.* 2014; Pressey 2014)

impacts (Russ *et al.* 2004), and do require the clear specification of the full counter-factual scenario.

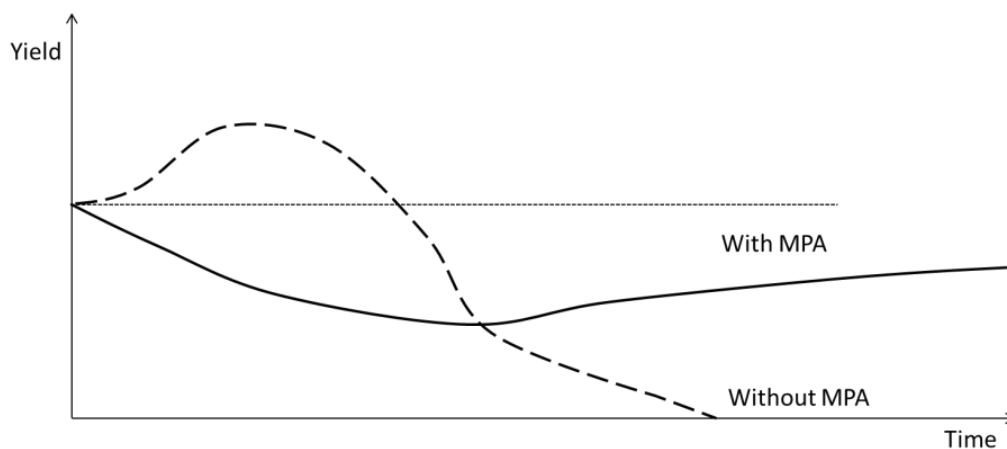


Figure C3 Hypothetical yield from an overharvested fishery with time both with and without an MPA where the absence of a counter-factual scenario changes the interpretation of MPA impact data

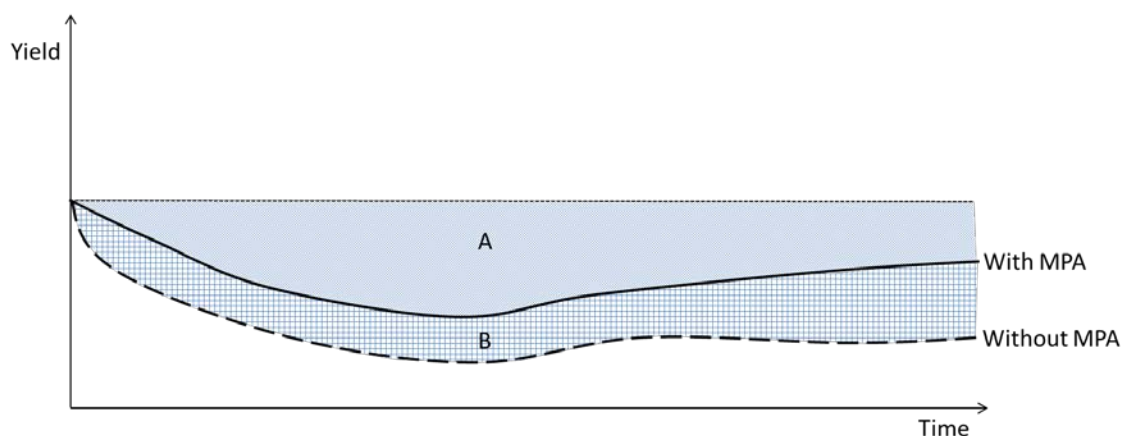


Figure C4 Alternative hypothetical yield from an overharvested fishery with time both with and without an MPA where the absence of a counter-factual scenario changes the interpretation of MPA impact data. Zone A represents the loss that would be perceived if the impacts of the MPA were measured relative to the baseline yield, whereas zone B represents the gain that would be perceived if the impacts of the MPA were measured relative to the counterfactual scenario without the MPA

The importance of specifying a counterfactual scenario, as discussed above, rather than just documenting a baseline starting point is illustrated by Dueri and Maury (2013). They used a numerical model of basin-scale population dynamics of skipjack tuna under environmental conditions and fisheries exploitation to test how large an MPA would need to be in order to have a positive effect on the tuna population and fisheries. They found the MPA in question had to be much larger than the Chagos MPA, and that even under these circumstances, *although the catch is higher than in the counterfactual scenario, it is still lower than in 2010*. Had the outcomes of the model

been compared just to the baseline starting point, the analysis would have appeared to show a negative impact on fisheries rather than a positive impact.

One of the relatively rare examples found where an empirical study considered baseline trends is (Castro *et al.* 2007). This study featured 3 years of time series data on large pelagic fish that preceded the designation of the Seaflower MPA. This data showed, on average, a progressive, pre-MPA reduction in mean CPUE over this time period, and also showed there was spatial heterogeneity in the baseline CPUE trends across different reefs within the study site (Castro *et al.* 2007). This implies that impact studies focused on the Seaflower MPA that ignore the pre-existing baseline fisheries trends would likely underestimate the true impact of the MPA on fisheries.

Confounding effects

There are a variety of confounding effects that can undermine the clear determination of MPA impacts. These confounding effects include, but are not limited to, habitat structure (Lozano-Montes *et al.* 2012; Stelzenmuller *et al.* 2009; Tupper and Rudd 2002), additional or altered regulation of fisheries and fisher behaviour (including the spatial distribution of their fishing effort) (Alcala *et al.* 2005; Eide 2012; Lozano-Montes *et al.* 2012; Parker *et al.* 2013; Pelletier and Mahevas 2005; Russ *et al.* 2004), environmental change unrelated to MPAs (Alcala *et al.* 2005; Beare *et al.* 2013; James *et al.* 2012; Mann and Pradervand 2007; Pastoors *et al.* 2000; Pistortus and Taylor 2009), and changes in other marine industries unrelated to the MPA but that overlap with MPA designation and implementation (Gomez *et al.* 2006).

The Apo Island MPA is a good case study to illustrate how regulatory and behavioural changes can obscure the impacts of MPA designation. Relatively long term data is available for the Apo Island MPA that measures CPUE (kg/person/hr) for hook and line fishing. This data shows an increase in CPUE since MPA designation, and this has been taken as evidence that spillover is occurring and benefiting fishery yield (Russ *et al.* 2004). However, over the same time period, overall effort expended on hook and line fishing declined by 46%, new legislation was passed that further restricted who can legally fish, and there was a dramatic increase eco-tourism as an alternative livelihood to fishing. Furthermore, fishermen distributed their effort differently across species with time following MPA designation. Consequently, the increase in CPUE cannot be attributed solely to the designation of the MPA and it cannot be taken in and of itself (and in isolation from other evidence) as indicating that the MPA is having a positive impact on yields. Without full consideration of these other changes, it will not be possible to isolate the impact of the MPA from the impact of the full suite of changes that has occurred around the island.

In the context of environmental change, the studies returned by the searches conducted for this report appeared to have a tendency to focus on (generally negative) environmental changes that can undermine MPA performance with respect to fisheries. In the St. Lucia Marine Reserve in South Africa, for example, there is evidence that the observed decline in stumpnose (*Rhabdosargus arba*) is not due to any particular feature of the MPA or its enforcement, but instead is a consequence of the closure of the mouth of the St. Lucia estuary (Mann and Pradervand 2007). Similarly, there is evidence that MPA designation may be unable to compensate for the negative effects on fish populations caused by hypoxia (Perez-Dominguez and Holt 2006). In the North Sea, despite decreased fishing effort in the "Plaice Box," morality has increased, and this mortality has been attributed to changes in the North Sea ecosystem starting in the 1990s (Beare *et al.* 2013; Pastoors *et al.* 2000). In Plettenberg Bay in South Africa, seal populations have recovered (in and of itself a positive change). However, the seal

populations now consume a greater quantity of sardines than to purse-seiners (Huisamen *et al.* 2012), and so have become a confounding factor obscuring the relationship between MPA protection and fisheries yields.

This range of examples serves to illustrate that there are a wide variety of factors that can affect the health of marine species and fisheries yields besides MPA designation, and that unless these factors are explicitly controlled for and documented in MPA impact case studies, it is difficult, if not impossible, to isolate the impact of the MPA from the impact of a suite of social and environmental changes. Consequently, studies may inappropriately either underestimate or overestimate the effects of MPAs in terms of fish yields.

Insufficient data on fished species

In a similar theme to the lack of baseline data and counterfactual scenarios, it is also the case that a lack of spatially-explicit data on species abundance, movement patterns, and catch in relation before and after MPA implementation undermines the ability of researchers to detect the impact of MPAs on spillover and yields (Kellner *et al.* 2007; Russ *et al.* 2004). This is particularly relevant in contexts where marine populations are heavily overfished prior to the designation of MPAs or a network of MPAs, and where MPAs may, consequently, be insufficient without additional and radical changes out with the MPA (Muallil *et al.* 2014). In this type of situation, in the absence of sufficient data on the stocks, it would likely be the case that MPAs would severely underperform relative to expectations that were set based on inaccurate assumptions regarding the true environmental baseline.

Although not necessarily feasible, ideally information would also be available on which locations are sources/sinks for the larvae and adults of populations MPAs are being designated to protect. The location of source/sink sites relative to the location of an MPA (or MPA network) is important because there is evidence (primarily from MPA modelling studies) that when MPAs are located primarily at sink sites rather than source sites (Gerber *et al.* 2003; Pelletier and Mahevas 2005), or at sites that are not connected by wind-driven advection to suitable sink sites (Hinrichsen *et al.* 2009), that populations (and by extension catch) may decline after MPA designation. This result has not been achieved universally within the MPA modelling literature (e.g. Levin and Stunz 2005). However, where studies have concluded it is important to protect the source sites, the reason that populations (and catch) may decline when MPAs are located at sink sites is as follows: MPA designation at sink sites may then concentrate fishing effort on source sites, thereby undermining future supply to sink sites (Pelletier and Mahevas 2005). Furthermore, populations may decline when MPAs are not well connected by advection to sink sites because it may undermine the success of larvae that are dispersed from any part of the adult population taking refuge within the MPA (Gerber *et al.* 2003). At least one non-MPA study has obtained results that corroborate this line of reasoning. Sundblad *et al.* (2014) mapped nursery areas in an archipelago of the Baltic Sea and concluded that the availability of nursery areas functions as a bottleneck, ultimately constraining adult population sizes.

In line with this idea that distinguishing between sources/sink sites is important to understanding the impact of MPAs on fisheries, (Pelletier and Mahevas 2005) caution against the use of meta population models that feature identical environmental patches. These models, by failing to acknowledge the difference between source/sink sites tend to demonstrate a positive impact on yields that are not realistic from the perspective of policy formation. Importantly, however, detecting source/sink sites and understanding the relationship between those sites, habitat characteristics, and larger

hydrodynamic forces may require that studies consider much wider spatial scales than might otherwise be deemed necessary (Etherington and Eggleston 2000).

Insufficient data on fishing activity: further consideration of CPUE

As mentioned in section 4.1.1, CPUE is measured as a common proxy for spillover in the context of assessing the impacts of MPAs on fishery yields. In addition to measuring discrete CPUE data points, some research (e.g. Follesa *et al.* 2011) also measures gradients of CPUE across MPA boundaries in order to provide an indication of whether or not spillover is occurring. This, however, is also a potentially poor proxy to use for MPA impacts, particularly in the absence of other data on fishing activity. One reason for this stems from that discussed in the preceding section – that the location of MPA(s) relative to source/sink sites and the hydrodynamic conditions between sites, affect MPA success. This means that CPUE figures could be misleading in the absence of information on whether the CPUE figures related to concentrated effort at a source site or not.

Another reason is that CPUE can be driven by factors that are not necessarily connected to MPAs. For example, Stelzenmuller *et al.* (2009) found that CPUE was only statistically significantly correlated with distance from *Posidonia oceanica* beds, and that it did not have a statistically significant relationship with any of the features of MPA in question, implying that there may be some instances when changes in CPUE are incorrectly attributed to MPAs. Additionally, there is some evidence that CPUE may be an unreliable proxy even for fisher welfare, as some research indicates that decreasing CPUE does not necessarily correlate with decreased wellbeing (McClanahan 2010).

Most importantly, however, is the limitation of focusing on CPUE with respect to understanding the impact of MPAs on yields. As Abesamis *et al.* (2006) argue, the magnitude of spillover (and the associated benefits both in terms of yield and financial value) cannot actually be estimated without information on total yield (and changes in total yield through time). This information was not often present in the studies found. A recently published meta-analysis, for example, considered 28 data sets from 7 MPAs in southern Europe and modelled a number of different relationships involving CPUE and other MPA features such as distance, area, duration of protection, and type of species (Vandeperre *et al.* 2011). However, this study did not include any analysis of total catch and so cannot contribute to assessment of the impact of MPAs on fisheries through spillover.

Consideration of total yield (and by extension total fishing intensity) is also important because of the potential for fishing intensity to undermine the persistence of populations within MPAs. Even when density-dependent spillover occurs, if the overall fishing intensity outside the MPA is too intense, it may become difficult to achieve a stable population size within the MPA itself or to maintain the expected species assemblages (Eide 2012; Freeman *et al.* 2009; Hobday *et al.* 2005; Lozano-Montes *et al.* 2012; White *et al.* 2013). This is particularly relevant if the spillover observed or measured is not truly permanent, and instead is bidirectional (movement that effectively increases the contact between populations in marine reserves and fishers) (Goni *et al.* 2006). Because individuals within an MPA have to reproduce before they are fished in order for the population to persist, and because emigration from MPAs may be bi-directional and have no fixed temporal relationship to reproduction, “fishing

the line" behaviour¹⁷ may exert strong pressures on populations within MPAs (Kellner *et al.* 2007). Hence, increasing CPUE analysed without reference to overall catch, overall effort, and biomass inside an MPA may signal the effective depletion of the MPA rather than net gains to fisheries due to MPA designation and enforcement.

Furthermore, there is a need to consider how many species are targeted in a particular fishery when fishers respond to MPA designation with "fishing the line" behaviour. The results of a theoretical modelling study (i.e. Kellner *et al.* 2007)¹⁸ show, for example, that fishing the line behaviour cannot be simultaneously optimally distributed for species with different mobility *rates*, even when spillover is occurring. This means that total yield will be affected not only by spillover and the location of fishers relative to the MPA, but also by the number of species fished and their mobility rates. This study also found that increased populations and catch only occurred within a single-species fishery when fishing effort was optimally distributed, the species in question has moderately high mobility rates, and the target species is overexploited prior to MPA designation (Kellner *et al.* 2007), implying (as have other studies, e.g. Miethe *et al.* 2009) that there are a wide range of scenarios where fisher behaviour and stock attributes both prior and subsequent to MPA designation may interact and contribute to a situation where yields do not increase post-MPA designation.

MPA impacts are multifaceted and interactive

As established in the preceding sections, the impacts of MPAs are multi-faceted. For example, habitat heterogeneity interacts with fish populations to impact on MPA effectiveness (Stelzenmuller *et al.* 2007). Similarly, it may be that MPAs impact on the variability in yield, rather than simply the total yield (Gerber *et al.* 2003). By extension, studies that try to determine the fisheries impacts of MPAs by only considering a few indicators are missing pieces of information that are analytically relevant to the assessment of the fisheries impacts of MPAs.

Although a detailed review of the MPA modelling efforts is beyond the scope of this study¹⁹, the potential impacts of failing to include analysis of all the relevant facets of MPAs is best illustrated by considering the outcomes of modelling studies that do consider a wide variety of facets of MPA impact. Ainsworth *et al.* (2012), for example, used the Atlantis Model to compare the impacts within the Gulf of California of various MPA scenarios with scenarios in which there was full enforcement of existing, but fishing regulations. The model featured 63 functional groups and simulations were run over time period of 25 years. By specifying this many functional groups, maintaining a long-term focus, and by being able to consider simulated counterfactual scenarios, the model was able to overcome some of the limitations often seen in empirical studies.

The results for a scenario in which 7 MPAs varying in size from 83km² – 17,596 km² were implemented as no take zones (and for which fishing effort was eliminated rather than displaced) were as follows: For the smaller MPAs, while the MPA designations led to increased catch for some trophic levels (i.e. tertiary consumers and primary producers), it also led to decreased catch for other trophic levels (i.e. secondary consumers and basal species). In total, these two effects nearly cancelled each other out, as the simulations resulted in a <0.5% net gain in yields across all trophic levels in

¹⁷ "Fishing the line" is where fishermen relocate fishing effort in response to MPA designation and the associated expectation that spillover will occur and that both catch will increase and the size of individuals caught

¹⁸ Note that this study employed several conservative assumptions: decreased fishing effort doesn't affect habitat quality; larval transport/production is ignored; fecundity is constant.

¹⁹ See: Pelletier and Mahevas (2005)

the small MPAs following MPA designation. For the larger MPAs, although catch of secondary and tertiary consumers increased, it did not increase enough to even compensate for the lost fishing area, resulting in a net decline in yields of as much as 19% (depending on the MPA considered) (Ainsworth *et al.* 2012).

The different scenarios modelled also highlighted that it is feasible for the commercial yield and recreational yield to respond differently to MPA designation and to the full enforcement of non-MPA regulations, with recreational fisheries tending to benefit in terms of catch. Interestingly, this study also presented some evidence that both MPAs and strict enforcement of non-MPA fisheries control measures can trigger trophic level cascades that actually undermine fisheries yields. The example given in this study featured crab populations that first increased after a decrease in fishing pressure and then declined following the increased crab predation that ensued after the population of crabs increased. In turn, this increased rate of natural crab predation led to decreased catch despite the sustained reduction in fishing effort (Ainsworth *et al.* 2012). As a case study, Ainsworth *et al.* (2012) effectively highlights the complexity linking MPA governance and fishery yields.

Similarly, Lozano-Montes *et al.* (2012) used an Ecospace model to consider the effects of the Jurien Bay Marine Park on commercial rock lobster fishery as well as the co-located recreational fishery. The simulations covered 20 years starting in 2007. This study found that different trophic levels responded differently to MPA implementation, and that MPA was most effective when overlapped with highly structured habitats (and was therefore less effective when overlapped with less structured habitats like sand flats and seagrass beds)

Although the above discussion refers to just two studies, when considered together, they serve to illustrate that there is at least some evidence that changes in yield post-MPA designation may vary with the trophic level considered, the nature of the fishery considered, and the stringency of enforcement. Consequently, studies that focus on just one aspect of MPA impact may miss important aspects of the relationship between MPAs and yield.

C3.1.4 A way forwards

Given all of the preceding discussion highlighting that the impacts of MPAs are heavily dependent on fine-scale site and species-specific factors (Tupper and Rudd 2002), and that there are a wide range of barriers that limit the extent to which especially the existing empirical literature on MPA impacts can be used to quantify a relationship between MPA designation and fishery yields (as mediated by spillover), the recommendation for this study is that the MPA-seafood relationship be investigated broadly using existing modelling studies. Such studies may provide useful insights while at least partially overcoming some of the barriers discussed previously. Even within the pool of available modelling studies, however, there is a lack of consensus regarding under what conditions MPAs should generate increased yields for fisheries and how they compare to alternative marine management policies (Perez-Ruzafa *et al.* 2008). Furthermore, very few models were identified in the time available for this review that could be ‘transferred’ in the form of relatively simple and accessible equations for use in the context of analysing the ecosystem service impacts of global, theoretical MPA expansion scenarios. That said, however, the following are worth consideration:

Bensenane et al (2013)

Bensenane *et al.* (2013) is a study seeking to identify a theoretical relationship between the proportion of a fishery that is protected and catch in order to estimate the optimal reserve size (in terms of the long run equilibrium fish catch out with the reserve). The study utilizes simulation models, assumes fish growth follows the logistic growth pattern of Lotka-Volterra predatory prey models, assumes that fish movement occurs over two time scales (a fast movement between sites and slow movement related to growth with time). The relevant equations from Bensenane *et al.* (2013) for this study are 11 and 12, reproduced here as figures Eq. C4 and Eq. C5, respectively.

$$Y^* = \frac{rc}{pq(1-s)} \left(1 - \frac{c}{pq(1-s)K} \right) \quad (\text{Eq. C4})$$

$$s^* = 1 - \frac{2c}{pqK} \quad (\text{Eq. C5})$$

In these equations, Y^* is the catch at equilibrium out with the reserve, r is the fish population growth rate, c is the cost per unit effort to fish, p is the price per unit effort of fish, s is the proportion of the fishery area contained within an MPA, K is the carrying capacity for the full area included in the model, q is the catchability coefficient,²⁰ and s^* is the optimal size of the marine reserve (i.e. the proportion of the fishery area that when designated as an MPA corresponds to the highest long run equilibrium fish catch). Bensenane *et al.* (2013) note that at s^* , Y^* greatly simplifies to the following (Eq. C6):

$$Y^* = \frac{rK}{4} \quad (\text{Eq. C6})$$

The relationship between equilibrium catch and reserve size has the following shape (Figure C5).

²⁰ This is defined in a different study explicitly as the “the proportion of the total stock caught by one unit of effort” (Perez-Ruzafa *et al.* 2008). This can vary with “gear efficiency, selectivity, habitat structure, fish behaviour, age of fish, time of day, season, etc” (Perez-Ruzafa *et al.* 2008).

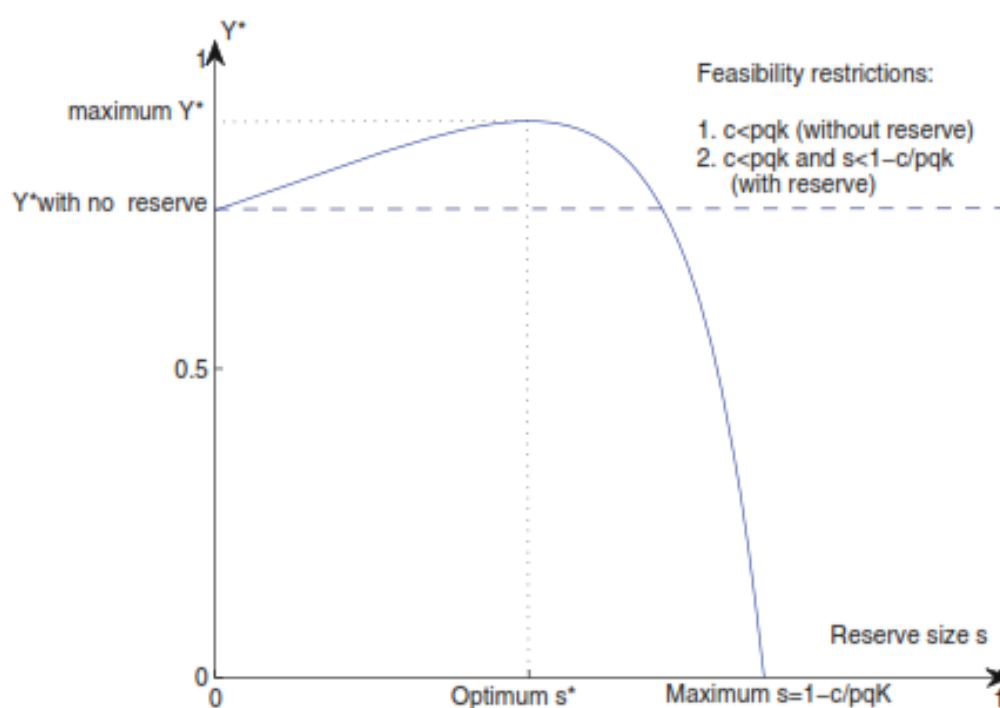


Figure C5 “The relationship between harvest and reserve size using [the] reduced system. Catches increased with MPA size reaching a maximum. The parameter values are: $r=0.9$, $c=0.6$, $p=1$, $q=0.5$, $K=4$...” Caption and figure from figure 2 from (Bensenane et al. 2013)

In terms of this study, the benefits of these equations are as follows: the relationships have been generalized and they rely on relatively few variables. If assumptions are made regarding growth rates, price, cost, carrying capacity, catchability, and proportion of fishery designated, then the catch associated with that designation can be estimated and compared to estimates of catch without an MPA designation.²¹

Perez-Ruzafa et al (2008)

This study conducted simulation analyses based on (instantaneous) logistic growth rates (r), harvesting rates (F), and diffusion coefficients (D , measured in length^2 per unit time) for individual species to simulate, and then estimate via multiple regression, the relationship between these three variables and flux of individuals across the MPA boundary (i.e. spillover). The outcome of this process (which was based on members of 7 families of fish) are shown below (Eq. C7, adjusted- $R^2=0.91$)

$$\text{Flux} = -5.35 + 0.01069D - 2.13 \times 10^2 + 1.02 \times 10^3 \quad (\text{Eq. C7})$$

The diffusion coefficient D can be estimated either based on the mean speed of a species when moving randomly and the mean free path seen in a species home range displacement, or can be estimated from the Einstein-Smolochowsky equation, while instantaneous fishing mortality (F) is a function of catchability and effort (i.e. $F_t = qE_t$) (Perez-Ruzafa et al. 2008). Although this equation does not actually model yields, has

²¹ The equation for catch at equilibrium without an MPA can be derived from Eq. 4 above by setting s (i.e. reserve proportion) equal to 0.

the advantage of not being reliant on a large number of variables, and it could be used in combination with a simplistic proportional assumption regarding the relationship between yield and spillover. Alternatively, Perez-Ruzafa *et al.* (2008) include an equation that can be re-arranged to relate total catch outside the MPA to the concentration of fish (n), their position in space (x), time (t), the diffusion coefficient (D), the logistic rate of growth (r), and the carrying capacity of the habitat (K) (Eq. C8):

$$Catch = \frac{\partial^2 n}{\partial x^2} - \frac{\partial n}{\partial t} + r \left(1 - \frac{n}{K}\right) n \quad (\text{Eq. C8})$$

Again, these equations have the benefit of already being generalized, and so may be useful in the context of this study.

Doyen and Bene (2003)

This study mathematically models the relationship between stock size, MPA size relative to fishing area, and guaranteed catch (C). The guaranteed catch function behaves as follows (Eq. C9):

$$C(A, N_0) = \begin{cases} N_0(\bar{u} - \sigma)A, & \text{if } N_0 \leq F(A, \bar{u} + \sigma) \\ F(A, \bar{u} + \sigma)(\bar{u} - \sigma)A, & \text{if } N_0 \geq F(A, \bar{u} + \sigma) \end{cases} \quad (\text{Eq. C9})$$

Where A is the area fished relative to the total area under management, N_0 is the initial stock, F is the stock biomass function, $\bar{u}$ is the target harvesting rate, and σ is the degree of uncertainty in harvesting in the present time period ($\sigma = 0$ corresponds to full certainty).

When the initial stock N_0 , is greater than the equilibrium biomass function (i.e. $F(A, \bar{u} + \sigma)$), then the fraction of the fishery that can remain open to fishing and that will maximize the minimum guaranteed sustainable catch is (Eq. C10):

$$A^*(\sigma) = \frac{\sqrt{c}-1}{(\bar{u}+\sigma)\sqrt{c}} \quad (\text{Eq. C10})$$

Here, c pertains to the degree of density dependence in the stock recruitment of a species. When the initial stock is less than the equilibrium stock, given a certain area in which the stock exists, a target harvesting rate, and uncertainty, the maximum guaranteed catches are associated with the fished area = 1. This is only sustainable under certain conditions (see corollary A.3²²), and implies that the impact of MPAs on catches depends on various scenario-specific features including safe minimum biomass level, the harvesting fraction area, the degree of density dependence, and uncertainty in harvesting.

These three studies present different alternatives to modelling the catch resulted from MPA designation using generalized mathematical relationships reliant on relatively few variables. Which one is ultimately most suited for use in the analysis of a global MPA expansion scenario will depend on the global fisheries data available and the variables for which the most robust assumptions can be made.

²² "Corollary A.3. Assume that $c(1 - \bar{u} - \sigma) > 1$. If $N_{min} \leq \left(\frac{c}{d}\right) - 1(d(1 - \bar{u} - \sigma))$, then sustainability always applies, namely for any $A \in [0,1]$, any $\sigma \geq 0$, we have $Inv(A, \sigma) = [N_{min}, +\infty]$."

C3.2 MPAs and tourism

Another potential ecosystem service impact of MPA designation is either increased or improved tourism opportunities. Furthermore, financing by tourism is considered to be one possible avenue for financing MPA management post designation, especially in developing countries (Gelcich *et al.* 2013). However, the searches conducted to try and identify studies that documented relationships between MPA designation and tourism yielded a pool of literature that instead focused primarily on the potential negative impacts of tourism on MPA conservation objectives, rather than on the impacts of MPAs on tourism. Although one study claimed that MPA designation could affect the rate of eco-tourism and wild-life tourism (but not mass tourism) (Micheli and Niccolini 2013), and another highlighted that tourism had changed post MPA implementation but without making an unequivocal link to the role the MPA played in *driving* that change (Qiu 2013), only one study was found that directly quantified a relationship between MPA designation and changes in tourist numbers or experiences.²³ Furthermore, no studies were found that focused on quantitatively tracking trends in tourism as an impact of MPA designation. In other words, no studies were found to have employed anything like a BACI design with respect to tourism.²⁴

What this means is that there was insufficient evidence available to enable the discrimination between the possibility, on the one hand, that the dominance in the identified literature of studies focused on the negative impacts of tourism on MPAs (rather than the behavioural responses of tourists to MPA designations) provides an unbiased indication that the anthropogenic pressure dimension of tourism is more important than the ecosystem service dimension of tourism, and the possibility on the other hand, that academic research has focused to date more frequently and to a greater extent on understanding the anthropogenic pressure dimension of tourism rather than the ecosystem service dimension of tourism. Consequently, although some key themes that emerged from this pool of literature (detailed below) that highlight the importance of pursuing an evidence-based (rather than assumption-based) analysis of the ecosystem service impacts of MPAs, these themes in and of themselves cannot be interpreted as signalling that there is *no* positive impact of MPAs on tourism. Rather, as is the case with the relationship between MPAs and fisheries yields, there is a clear need to more carefully (and quantitatively) investigate the impact of MPA designation on tourism.

C3.2.1 Theme 1: Certain recreational activities may, in certain contexts, have a neutral impact on the marine environment

A limited pool of literature was found that highlighted instances where recreational activities had resulted in a neutral (or non-negative) impact on the marine environments within MPAs (at least depending on who is involved in the operation of tourist enterprises. See: Biggs *et al.* (2012)). There is some evidence that the Tavolara-Punta Coda Cavallo MPA in Sardinia has not been undermined by the continuation of multiple human activities within the MPA (Micheli and Niccolini 2013). This outcome was achieved, Micheli and Niccolini (2013) argue because of key individuals who actively fostered collaboration between, and worked with, the various users of the

²³ According to Brock and Culhane (2004) since the establishment of the Dry Tortugas National Park in 1992, visitation has increased 400% and boat registration has increased 50%. It is unclear the extent to which these changes were caused by the national park designation.

²⁴ This is despite Thurstan *et al.* (2012) evaluating the impact of non-consumptive recreational activities undertaking within 91 MPAs around the world with respect to the risk of those activities to the marine environment

marine environment included in the MPA. Their research also highlights the importance of including information on MPAs within the local educational curricula. Other research has found that experienced SCUBA divers in small groups who intentionally behave according to the precautionary principle may not negatively impact on the fish spawning aggregations they want to observe (Heyman *et al.* 2010). In agreement with this research, there is some evidence that at least with respect to the health of Mediterranean *Posidonia oceanica* meadows, sunbathing, swimming, snorkelling, and SCUBA diving may be pursued without incurring a negative impact (Lloret *et al.* 2008). Finally, the evidence is currently equivocal as to whether or not shark feeding alters shark behaviour or threatens metabolic rates (Fitzpatrick *et al.* 2011; Maljkovic and Cote 2011), though the evidence may be less equivocal in the context of reef fish (Ilarri *et al.* 2008).

C3.2.2 Theme 2: Tourism can directly conflict with the conservation objectives of MPAs

There are a wide range of examples drawn from a wide variety of recreational activities that illustrate the notion that tourism can be a threat to the “natural integrity” of the marine ecosystems contained within MPAs (Edgar *et al.* 2010), from the simple trampling of benthic assemblages (e.g. Casu *et al.* 2006; Juhasz *et al.* 2010) to boating (Burgin and Hardiman 2011; Manning *et al.* 2012), recreational fishing (Frisch *et al.* 2008; Rife *et al.* 2013; Westera *et al.* 2003), SCUBA diving, and participation in eco-tourism ventures.

In the case of recreational fishing, there is evidence, for example, that when it is allowed within an MPA, recreational fishing can function as the primary source of mortality of, and the most significant pressure exerted on, the species living within the MPA (Schroeder and Love 2002), thereby undermining population recovery. This has been found to be the case with the spiny lobster (*J. Edwardii*) in New Zealand (Shears *et al.* 2006), and may be especially true in the case of species that are attractive to recreational fishermen and that demonstrate high site fidelity (Blyth-Skyrme *et al.* 2006), or in the case of species for which there was a history of intense commercial fishing pressure prior to MPA designation (Diogo and Pereira 2014). Recreational fishing has also been documented as undermining the recovery of snapper within the Mimiwhangata marine park in New Zealand (Denny and Babcock 2004), and the recovery of mussel beds in the Ligurian Sea (Parravicini *et al.* 2010).

The impact of recreational fishing also connects, to a certain extent, to the idea that there is a host of confounding factors that affect the relationship between MPA designation and commercial yields (section C3.1.1), as there is some evidence that recreational fishing inside an MPA can directly compete with artisanal and/or commercial fishing outside that MPA. This competition for fish occurs when recreational fishers (including spear fishers) are allowed to extract and keep the fish caught, as this can end up reducing the catch that would otherwise be achievable by artisanal fishers outside the MPA (Albouy *et al.* 2010; Lindfield *et al.* 2014; Quach Thi Khanh and Flaaten 2010; Rocklin *et al.* 2011). An important implication of all these studies is that recreational fishing should be assumed to have a trivial or neutral impact on either conservation or fishery objectives.

SCUBA diving and snorkelling have also been found to promote negative impacts on marine ecosystems (and particularly reefs²⁵) (Lucrezi *et al.* 2013a, b; Silva *et al.* 2012). This damage may come in the form of contact-related damage,²⁶ anchoring, photography, sedimentation, or water pollution (Lucrezi *et al.* 2013b; Qiu 2013). Even just the operation of motor boats within an MPA may undermine the conservation objectives of that MPA. Research has found, for example, that boat noise provokes avoidance behaviour that effectively modifies the foraging and ranging behaviour of some fish species, such as *C. chromis* (Bracciali *et al.* 2012; Picciulin *et al.* 2010), and the feeding behaviour of some bird species, such as shags (*Phalacrocorax aristotelis*) (Velando and Munilla 2011), and even larvae (Holles *et al.* 2013). There is also some evidence that eco-tourism efforts can undermine the health of the marine ecosystem. For example, sea turtle watching in Greece has been documented as exerting pressure on sea turtle breeding areas (Schofield *et al.* 2013), and (Landry and Taggart 2010) suggest guidelines for sea turtle ecotourism guidelines to avoid the disruption of turtle metabolic patterns. Eco-tourism has also been documented as undermining the health of turtle grass (i.e. it is sparser, shorter, slower growing, and burdened with more epiphytes) in the Mexican Caribbean (Herrera-Silveira *et al.* 2010).

It is highly variable across specific case studies what drives these negative impacts. In some cases it may be that individuals do not perceive a certain action, such as close up photography (see: Lucrezi *et al.* 2013b) to be damaging, where as in other cases it may be a consequence of the scale of activities e.g. rapid expansion of activities. See: Qiu (2013), or even a lack of awareness by tourists that they are even in an MPA or a particular part of an MPA (e.g. Petrosillo *et al.* 2007; Smallwood and Beckley 2012). It may also, of course, be due to a consequence of ineffectual regulations, monitoring, or enforcement. The purpose of including this theme in this report is not to argue that negative impacts such as the ones mentioned here necessarily *always* happen,²⁷ but to highlight that there is clearly evidence that they *can* happen, regardless of whether tourism numbers actually increase post MPA designation. This means that it should not be uncritically assumed in the analysis of the benefits of MPA designation that the conservation objectives of the MPAs and the tourism that may be promoted as a means of financing the MPA are compatible with one another.

C3.2.3 Theme 3: Tourism can indirectly conflict with the conservation objectives of MPAs

A more limited set of studies were found arguing that tourism could have an indirect effect on the performance of MPAs. For example, Micheli and Niccolini (2013) argue that increasing coastal tourism (MPA-related to not) can lead to an increased demand for coastal infrastructure, the supply of which exerts pressures on marine ecosystems, including MPAs. In quite a different example, Milazzo *et al.* (2006) argued that tourist activities such as fish feeding can undermine the conservation objectives of MPAs by altering species behaviour, which in turn can trigger other changes in the ecology of the populations within the MPA (Milazzo *et al.* 2006). Finally, a couple of studies were

²⁵ Silva *et al.* (2012) compared two coral reefs in northeast Brazil and found that the reef with intense tourism scored worse than the reef without intense tourism for each of the following metrics: biomass, species richness, species diversity, and species dominance.

²⁶ For example, in California 65,000 person-days of recreational SCUBA diving were found to have caused the shedding of 130,000 blades of kelp (Schaeffer *et al.* 1999)

²⁷ Indeed, there is some evidence that there are ways of managing this impact, particularly with snorkeling and diving. Claudet *et al.* (2010), for example, found there was no evidence that snorkelers impact on local environments when they followed a trail laid out specifically to facilitate this activity within an MPA.

found that discussed the potential indirect impacts that recreational boating can have on nekton. (Bishop 2008) found that recreational boat traffic (over seagrass beds) decreases the populations of macro invertebrates (e.g. amphipods and polychaetes) that inhabit seagrass blades, and that in turn contribute to supporting the fish populations within the seagrass beds. Burfeind and Stunz (2007) found that the scarring (on greater than 15% of the seagrass beds) by boats undermines both the abundance and growth rates of White shrimp. The implication of these final two studies is that even the pursuit of non-extractive recreational activities may undermine the conservation objectives of MPAs if those recreational activities are boat-based.²⁸

C3.2.4 Theme 4: Tourism benefits may not always be perceived by local communities

There was some literature that focused on how local communities (particularly in developing countries) have perceived the impacts of the designation of MPAs. A good example of this is Bennett and Dearden (2014), a study that focused on 17 national MPAs on the Andaman Coast of Thailand and where individuals who are dependent on the marine environment indicated that the MPAs had not led to the expected benefits in the form of fisheries or tourism, but had undermined access to important cultural sites. This study signals that there is at least the perception of nontrivial trade-off between conservation objectives and livelihoods and community well-being, and further highlights the need not only to document MPA impacts clearly.

C3.2.5 Theme 5: There may be limits to the extent to which tourism can expand post MPA designation

An interesting notion was raised in a few studies that highlighted there may be limits to the extent that tourism can actually expand following MPA designation. These limits are not ecological in nature and result from the perception by tourists of the impact that other tourists have on their experience. For example, people have been shown to be sensitive to perceived crowding and by extension supportive of regulation that limits site crowding (Bell *et al.* 2011; Davis and Tisdell 1995; de Souza Filho *et al.* 2011; Inglis *et al.* 1999; Needham *et al.* 2011). This has also been found to be true in at least one site for activities that take place within the water. Roman *et al.* (2007) found, for example, preferences for fewer than 30 snorkelers per day per site within the Koh Chang National Marine Park in Thailand. Studies like these are suggestive of a kind of artificial, tourist preference-driven tourist carrying capacity within MPAs that may limit both the extent to which tourism expands most MPA designation (as people choose where to participate in their preferred tourism), and by extension the financial benefits that may come with MPA-related tourism. Interestingly, however, this social carrying capacity may not be constant, and may depend on the depth of environmental knowledge and recreational experience of the tourists in question (Inglis *et al.* 1999; Leujak and Ormond 2007).

²⁸ Other studies did not find this same effect of scarring, but cannot rule out its occurrence at higher levels of damage than those considered (e.g. Burfeind and Stunz 2006)

C3.2.6 Theme 6: Coastal tourism is not necessarily focused on marine ecosystem health

Implicit in some of the *ex-ante* anticipation that tourism will increase following MPA designation is the assumption that tourism is sensitive/elastic to improvements in marine ecosystems. For certain types of activities, this is almost certainly true: healthy reefs should be more attractive to SCUBA divers, for example, than depleted or damaged reefs. However, it is important to note that experiencing a healthy marine environment is not always the end goal of tourist activities. A recently published study demonstrated that some of the primary sources of meaning ascribed to Ningaloo Marine Park in Australia had nothing to do with ecosystem health, and instead had to do with things like spending time with family, escaping everyday life, and participating in diverse recreation away from home and an urban environment (Tonge *et al.* 2013). Tourism that is so driven may be relatively insensitive to ecosystem health, or to improvements that may result from MPA designation.

C3.2.7 Conclusions

The literature search did not yield any studies that could be used to quantify the impact of MPAs on tourism, and instead yielded literature that highlighted a number of ways in which the relationship between MPAs and tourism may not be particularly positive. Although this is not likely to universally be the case, and although the specific studies chosen here to illustrate this are limited in number and not exhaustive, the themes presented above do highlight the importance of acknowledging that the relationship between MPA designation, as mediated by social, political, and legal institutions, is sufficiently complex that it should not be assumed (in the absence of other evidence) either that designating an MPA will yield tourism benefits or that tourism will have a neutral impact on the marine environment in question.

Habitats and ecosystem services

The second set of results presented here focuses on the outcomes of the literature searches that targeted the literature on specific habitats (as opposed to the MPA literature). The habitats considered are: seagrass beds, macroalgae, mangrove forests, and coral reefs. The logic behind these searches was that MPA designation may, by protecting certain habitats, enable those habitats to provide ESs. The search strings used are included in Table C1.1. As this table shows, searches were conducted for a broad range of habitats using a variety of terminology for the relevant habitats (e.g. searches were conducted with both kelp and macroalgae). This section of the report focuses only on the information found regarding the relationships between specific habitats and ecosystem services, primarily supported by the habitats-specific searches but also augmented by relevant studies returned by the searches conducted in the MPA literature. This section is organized according firstly by ecosystem service and secondarily by habitat type. Information was not found for all habitat ecosystem service combinations, so some of the sections presented below are shorter than others.

C3.3 Climate regulation

C3.3.1 Seagrass

Seagrass beds contribute to the provision of the climate regulation service by sequestering carbon into plant tissues (i.e. shoots and roots). In at least some places (e.g. Japan) this is even true in coastal waters with shallowly submerged seagrass beds that are normally assumed to be sources of CO₂ (Tokoro *et al.* 2014). The literature returned by the searches conducted reveals a wide range of estimates for the quantity of carbon sequestered by seagrass beds, some of which is focused on quite localized case studies (e.g. Chiu *et al.* 2013; Dauby *et al.* 1995; Gacia *et al.* 2002; Greiner *et al.* 2013; Mateo and Romero 1997; Mateo *et al.* 2003; Pergent *et al.* 1997), and some of which are potentially relevant for this study because of their global focus (Table C3). The figures vary not only by the study site, but also the species, the density of the seagrass beds, the part of the plant considered, and the age of the seagrass beds. A relatively recently published study has estimated a global mean value, however, of between 41 and 66 gCm⁻²yr⁻¹ (Kennedy *et al.* 2010). In the event that the designation of MPAs allows for the recovery or expansion of seagrass beds²⁹, it would be reasonable in the context of the global MPA scenarios of this project to use this global average to provide a starting estimate for the resulting provision of the climate regulation service.

That said, for the sake of completeness, it is also worth highlighting that this carbon sequestration is not a guaranteed by-product of the existence of seagrass meadows. This is worth considering when one is working with scenarios that have a higher spatial resolution than the scenarios of this project. The amount of carbon sequestered varies with the species and age of the seagrass bed considered (Cebrian *et al.* 2000). Damaged seagrass beds sometimes release carbon dioxide back to the atmosphere (Macreadie *et al.* 2014). Increased grazing by species such as turtles (which may be protected as a consequence of MPA designation) can dramatically undermine the quantity of carbon sequestered from seagrass beds (Kelkar *et al.* 2013), as can increased sea surface temperatures (SST) (Pedersen *et al.* 2011), and anthropogenic activities (such as fish farming) that trigger a nutrient enrichment-driven shift from autotrophy to heterotrophy (Apostolaki *et al.* 2011). Furthermore, not all of the carbon that is fixed by seagrasses ends up being permanently sequestered – some is remineralised after being consumed by other marine organisms (microbial or larger) (Chiu *et al.* 2013; Pergent *et al.* 1997), and some may end up washed ashore in ‘banquettes’ that also can decompose over relatively short time scales and so cannot be considered to be permanently sequestered (Mateo *et al.* 2003).

²⁹ Something that some research would suggest is not likely to occur on human time scales, at least in the case of *Posidonia oceanica* meadows. See: Boudouresque *et al.* (2009).

Table C4 Estimates of carbon sequestration in seagrass beds around the world

Location	Species	Quantity (gCm ⁻² yr ⁻¹)	Type of Measure	Features	Source
Global	19 + mixed communities	41-66	Buried (globally) that originates from seagrass production	Looked at data on carbon sequestration in seagrass meadows from 88 places in world	Kennedy <i>et al.</i> (2010)
		48-112 (Tg C yr ⁻¹)	Buried (globally) inclusive of sediment trapped by seagrass beds		
		4.2-8.4 Pg C	Global pool of organic carbon contained in/by seagrass beds		
Global	Mixed (un-specified)	299 Tg C yr ⁻¹	The amount of carbon that could be released per year if all the organic carbon in the top meter of sediment was remineralised	This study compiled published and unpublished data on the organic carbon content of seagrass biomass and soils in 946 distinct seagrass beds	Fourqurean <i>et al.</i> (2012)
Global	<i>Zostera marina</i>	1.2-1.5 t C yr ⁻¹ km ⁻² (associated with 31 t gross production of seagrass per square km per year)	Amount of carbon out of the 31 tC gross production of seagrass per km ² that ends up being trapped in deep water after 100 years	This carbon is effectively sequestered because it is trapped in deep water masses	Suzuki <i>et al.</i> (2003)

C3.3.2 Macroalgae

Macroalgae stands can contribute to the provision of the climate regulation service in much the same way as can seagrass beds: by sequestering carbon. The literature returned contained a range of estimates related to individual sites (e.g. Corey *et al.* 2012; Wada *et al.* 2008; Zhou *et al.* 2006), but also featured some research that highlighted that macroalgae often supports secondary production (up to several kilometres away from the source stands) (Kelly *et al.* 2012; Krumhansl and Scheibling 2012). This implies that the production of detritus may not be a reasonable estimate of the quantity of carbon actually *sequestered* as a consequence of kelp stand growth. Therefore, although there are some estimates that could, in theory, be used to help estimate changes in the provision of the climate regulation service as a consequence of changes in the extent of macroalgae resulting from MPA designation (such as the global average production associated productivity reported in (Krumhansl and Scheibling 2012) (Table C4), there would be significant uncertainty associated with any estimates resulting from the application of the aforementioned figures.

Table C5 Estimates of carbon sequestration in kelp stands

Location	Species	Quantity	Type of Measure	Features	Source
Global Average	??	706 gCm ⁻² yr ⁻¹	This figure is equivalent to 82% of the average global productivity of kelp	Kelp is also responsible for secondary production offshore from the kelp stand	Krumhansl and Scheibling (2012)
Southern Korea	Ecklonia (Brown algae)	10 tonnes of CO ₂ ha ⁻¹ yr ⁻¹	This is the amount of carbon sequestered in a year in the context of an algae farm	This study also found that the production of this algae also reduced the dissolved inorganic carbon concentrations within the water column	Chung, Oak, <i>et al.</i> (2013)
Coastal territory from Vancouver to the Aleutian Islands	Order Laminariales	(1) 313-900 gCm ⁻² yr ⁻¹ (2) 25-70 gCm ⁻² yr ⁻¹ (3) 4.4-8.7 Tgyr ⁻¹	Net primary productivity (NPP) produced by kelp stands in the presence (1) and absence (2) of urchin-consuming sea otters, and (3) total kelp-driven carbon sequestration attributable to sea otters	In controlling the urchin populations, otters facilitate significantly more carbon sequestration by kelp stands than would otherwise be possible.	Wilmers <i>et al.</i> (2012)

C3.3.3 Mangroves

Mangroves can contribute to the provision of the climate regulation service primarily through the sequestration of carbon in plant material and the export of carbon from coastal systems in to deeper water systems. The literature search returned a wide variety of estimates related to carbon sequestration in particular mangroves forests (e.g. Adame *et al.* 2014; Alongi *et al.* 1998; Ceron *et al.* 2011; DelVecchia *et al.* 2014; Duarte and Cebrian 1996; Gladstone-Gallagher *et al.* 2014; Hossain 2014; Leopold *et al.* 2013), and some global synthesis research (Table C5). Of particular note in this pool of literature are the global studies shown in the first several rows of the table. These studies will likely be of the most use to examining the ecosystem service impacts of MPA scenarios that lead to the expansion of mangroves forests. It is important to note that two (and fairly recently published) of these studies (e.g. Alongi 2012; Breithaupt *et al.* 2012) provide estimates that are fairly similar to each other: 163 g organic C m⁻² yr⁻¹ and 174 g C m⁻² yr⁻¹, respectively. Consequently, this range could be used in the analysis of an MPA scenario that was expected to result in increased mangrove area as a consequence of MPA designation, although it should be acknowledged that there are bound to be high levels of uncertainty associated with these estimates (Hopkinson *et al.* 2012).³⁰

³⁰ That said, it is worth noting that some researchers have argued that estimates of carbon storage and storage rates cannot be scaled up from site-specific values to regional values unless the drivers of variability across the region are known (Saintilan *et al.* 2013). This is rather supported by other recent research that highlights that carbon sequestration is a highly context-dependent process and that it cannot be estimated from the more easily observable above ground parameters (DelVecchia *et al.* 2014).

The literature also returned a number of studies that were focused on the preservation of existing carbon stocks in marine sediments (e.g. Adame *et al.* 2013; Alongi *et al.* 2012; Kauffman *et al.* 2014; Lovelock *et al.* 2011; Pendleton *et al.* 2012; Wang *et al.* 2013). These studies emphasized that if mangroves are degraded or cleared that there is the potential for very substantial emissions of carbon dioxide into the atmosphere from marine sediments. Of particular interest to this study are Kauffman *et al.* (2014) and Lovelock *et al.* (2011) as these studies provide quantitative estimates of these emissions. The former estimates that clearing one hectare of mangroves and converting it a shrimp farm would release 2244-3799 Mg CO₂eq per year, while the later estimates that the annual average emissions across a 20 year time period of clearing one square kilometre of mangrove is 3,000 tonnes of CO₂ per year.³¹ These figures could be relevant to the current study in that they may provide a means of estimating avoided ecosystem service losses relative to a counterfactual scenario where mangrove forests are cleared rather than protected.

In addition to noting these global average figures, it is worth noting a little more detail on carbon sequestration in mangrove forests as well as a number of nuances and caveats. Firstly, carbon sequestration has been found to be positively correlated with factors such: the age of the site, tree height, tree diameter, net canopy photosynthesis, above ground biomass (AGB) belowground biomass (BBG), total biomass, carbon stock, growth efficiency, the ratio of AGB to tree height, tree girth, leaf area index, and silt content. Conversely, carbon sequestration is negatively correlated with soil temperature and sediment clay content (Kathiresan *et al.* 2013). Carbon sequestration in mangrove forests is also affected by salinity and inundation (i.e. tidal) patterns (Alongi 2011; Barr *et al.* 2010; Zhang *et al.* 2013) and sedimentation patterns (Yang *et al.* 2014). Some research argues that as a consequence of this, mangrove restoration must also endeavour to “recover” the hydraulic conditions associated with mangrove forests if the restoration is going to restore the carbon sequestration capability of a restored mangrove forest (Matsui *et al.* 2012). Carbon sequestration rates are further affected by the specific plan community in question, and background local sedimentation rates (Lovelock *et al.* 2014), as well as the level of disturbance experienced to date (Howe *et al.* 2009), and the level of nutrient enrichment experienced by the forest (i.e. nutrient enrichment has been shown to increase carbon sequestration in at least a few systems. See: Keuskamp *et al.* 2013; Sanders *et al.* 2014).

Secondly, it is important to recognize that not all mangroves, as non-linear, non-equilibrium systems, sequester carbon (Alongi 2011), and that not all of the carbon that is fixed by mangrove forests that do sequester carbon is actually sequestered. Firstly, carbon will only be sequestered in sediments if those sediments are either derived *in situ* or if the carbon would not otherwise have been sequestered (had the sediment not been trapped in the mangrove AGB (Saintilan *et al.* 2013). Secondly, mangrove forest export both dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) to surrounding environments, with the export of DOC constituting as much as 10% of the global terrestrial flux of DOC to coastal ecosystems (Bergamaschi *et al.* 2012; Bouillon *et al.* 2008; Miyajima *et al.* 2009), as well as in the form of litter and particulate organic carbon (POC) (Adame and Lovelock 2011; Machiwa and Hallberg 2002). As DIC, DOC, POC, and litter are all available to support secondary production in neighbouring ecosystems, the volumes of carbon exported in these forms will not all be formally sequestered. Litter and POC may also be mineralized by microbial communities (Kathiresan *et al.* 2013). This mineralization is slower in

³¹ It is important to note that the clearing of mangroves may also result in emissions of methane and nitrous oxides (Konnerup *et al.* 2014).

sediments than in the water column, in dense structures (like wood as compared to leaves), and in temperate areas as opposed to in the tropics (Gladstone-Gallagher *et al.* 2014).

Overall, therefore, although there are global estimates related both to mangrove carbon sequestration rates and CO₂ efflux rates associated with mangrove clearing, and although these estimates could be used within the context of global MPA expansion scenarios, they should be used with the caveats that there is high uncertainty associated with those averages not in the least because carbon sequestration is affected by a wide range of factors in mangroves, making it a highly site-specific features of mangrove forests.

Table C6 *Estimates of carbon sequestration in mangrove forests around the world*

Location	Species	Quantity	Type of Measure	Features	Source
Global	Mean across all species	24 TgCyr ⁻¹	Mangroves occupy 0.5% of the coastal area (globally) and contribute this amount to carbon storage each year.	This represents 10-15% of coastal sediment carbon storage annually.	Alongi (2014)
Global	Mean across all species	90-970 Tg C y ⁻¹	Potential emissions from mangrove forest deforestation	The potential emissions can exceed the storage capacity of these forests	Breithaupt <i>et al.</i> (2012)
Global data	Does not specify (Unspecified all species)	163 g OC m ⁻² yr ⁻¹ 95% CI: (132, 204)	This study estimated the mean annual rate of organic carbon burial in mangroves and the total global organic carbon burial in mangroves	This study estimated geometric means, and concludes that over a century, 8-15% of all organic carbon burial in marine systems occurs within mangroves. It also features the results from 7 previous reviews on this subject should the reader be so interested	
		26.1 Tg OC yr ⁻¹ 95% CI: (21, 32.4)			
Global data	Unspecified species mix	174 g C m ⁻² yr ⁻¹	Average carbon burial in mangroves	Most of the carbon stored by mangroves is stored in sediments and dead roots. The quantity stored is approximately 14% of the carbon stored within the global oceans	Alongi (2012)
Global data	Unspecified species mix	218 +/- 72 Tg C yr ⁻¹	Global primary production in mangroves	This study concludes that rates of mineralization of carbon in mangroves systems and the export of carbon in dissolved inorganic form are severely underestimated, as is the efflux of CO ₂ from sediments	Bouillon <i>et al.</i> (2008)
Global data	Unspecified species mix	112 +/- 85 Tg C yr ⁻¹	Amount of carbon that is fixed by mangroves that is unaccounted for in existing estimates of carbon fluxes		Wang <i>et al.</i> (2013)
Yingluo Bay, Guangdong Province (South China)	<i>Avicennia marina</i> :	212.88 t ha ⁻¹	These are the carbon stocks associated with different mangrove species as measured within the top 50cm of sediment	This provides some indication of the magnitude of the carbon that could be potentially released if these mangroves were cleared	
	<i>Sonneratia apetala</i> :	262.03 t ha ⁻¹			
	<i>Aegiceras corniculatum</i> + <i>Kandelia obovata</i> :	323.57 t ha ⁻¹			
	<i>Rhizophora stylosea</i> :	443.13 t ha ⁻¹			
	<i>Bruguiera gynmnorhiza</i> :	376.80 t ha ⁻¹			

Table C7 Estimates of carbon sequestration in mangrove forests around the world (continued)

Location	Species	Quantity	Type of Measure	Features	Source
Caribbean	Mix of shrub, medium, and tall mangrove species	2244-3799 Mg CO ₂ eq ha ⁻¹	This is the potential emissions associated with converting 1 ha of mangroves to a shrimp pond	This shows there is a huge potential disservice that may be realized if mangroves clearances continue	Kauffman <i>et al.</i> (2014)
Caribbean	Unspecified species mix	10,600 t km ⁻²	Efflux of CO ₂ in the first year after mangroves are cleared	Annual efflux of CO ₂ following mangroves clearances decreases with time	Lovelock <i>et al.</i> (2011)

C3.3.4 Coral reefs

None found.

C3.4 Erosion prevention

C3.4.1 Seagrass

Seagrass beds can contribute to the provision of the erosion prevention service by potentially reducing the erosive power of waves and currents (i.e. by reducing wave height, wave velocity, and current velocity, and by changing wave and current patterns), and by trapping sediment locally.³² The literature returned by the searches conducted to locate evidence related to the relationship between seagrass and erosion prevention revealed a mix of lab-based studies, field-based studies, theory-based studies, and model-based studies (e.g. Backhaus and Verduin 2008; Blackmar *et al.* 2014; Bradley and Houser 2009; Chen and Zhao 2012; Chen *et al.* 2007; Elginos *et al.* 2011; Infantes *et al.* 2012; Luhar *et al.* 2010; Luhar *et al.* 2013; Maza *et al.* 2013; Mendez and Losada 2004; Moller *et al.* 1999; Paul and Amos 2011; Paul *et al.* 2012; Peterson *et al.* 2004; Pinsky *et al.* 2013; Pujol *et al.* 2013; Stratigaki *et al.* 2011; Verduin and Backhaus 2000; Yang 1998).

This collection of studies did provide broad support to the idea that seagrass beds provide this service. However, it is worth noting that it is necessary to recognize a number of nuances with regards to the link between seagrass beds and the prevention of coastal erosion. Firstly, the degree of current attenuation depends on the type of seagrass because the different morphological structures found in different species (Backhaus and Verduin 2008). Secondly, the degree of current attenuation depends on the type of current: the impact on oscillatory velocities typically is less than the effect on unidirectional flows (Luhar *et al.* 2010; Luhar *et al.* 2013), and tidal currents were found in one study to reduce the wave attenuation capacity of seagrass beds (Paul *et al.* 2012). Thirdly, the impact of seagrass beds on wave attenuation and wave height depends on the frequency of the waves in question, as there may be some frequencies where the seagrass beds do not provide any wave attenuation (Bradley and Houser

³² Within the Mediterranean, there is also evidence that the banquettes formed by *Posidonia oceanica* fronds and that subsequently wash up on beaches play an important role in maintaining beach morphology (Daby 2003; De Falco *et al.* 2008; Simeone and De Falco 2012). De Falco *et al.* (2008) estimated that each cubic meter of *Posidonia* banquette contains 92.8 kg of sediment (on average), and that by extension the removal of 106,180 m³ of *Posidonia* from 44 beaches in Sardinia removed a substantial quantity of structural beach material as well.

2009). Finally, there are also certain combinations of seagrass density and current velocity that may increase localized current speeds due to the increasing impenetrability of the seagrass bed to water flow (Backhaus and Verduin 2008). These nuances support the notion highlighted in recently published research that the efficiency of the provision of this service is affected by the energy flux in the environment, the density of shoots, the magnitude of standing biomass, and plant stiffness, and other morphological features, and that the highest level of provision will come from large, long-lived, slow-growing species with high, seasonally-constant biomass (Elginos *et al.* 2011; Ondiviela *et al.* 2014).

Interestingly, although a variety of relationships between the relevant variables were specified in the literature found, the studies found did not, by and large, go further to emphasize the consequences of reduced wave and current attenuation in terms of sedimentation and/or erosion. Thus, although this literature can serve as evidence that seagrass beds can often provide the erosion prevention service, and provides information on what variables may increase or decrease the provision of the service, this pool of literature cannot easily be used to quantify changes in the provision of this service as a result of any policy (e.g. MPA designation) that may affect seagrass bed health and/or extent (as this would need to be measured in terms of changes in sedimentation). One possible exception to this trend is Chen *et al.* (2007), a study that explicitly models sediment transport in response to changes in seagrass in Maryland. The models utilized in Chen *et al.* (2007) are likely not going to be practical for use in a global MPA expansion scenario analysis, however, as they require depth-averaged velocity information as well as diffusion coefficients.³³ Another exception to this was as study returned that actually related to saltmarsh stands rather than seagrass beds that estimated a sedimentation rate of 298 gm⁻² of *Scirpus marigueter* (Yang 1998). Given the improvements in modelling wave and current attenuation, a fruitful avenue for future research would be the expansion of these models to consider rates of erosion directly in specific contexts.

C3.4.2 Macroalgae

Macroalgae stands can contribute to the provision of the erosion prevention service in much the same way as can seagrass beds. However, as with seagrass beds, the role of macroalgae stands in attenuating waves is not uncontroversial. As Pinsky *et al.* (2013) highlights, wave attenuation is driven by the interaction of geomorphic, ecological, and hydrodynamic factors. As was the case with the seagrass studies, the macroalgae studies highlight literature focused on the wave attenuation rather than the associated effects in terms of net sedimentation and erosion. Consequently, although there is evidence that macroalgae can (at least under certain circumstances) contribute to the provision of the erosion prevention service (e.g. Andersen *et al.* 1996; de Bettignies *et al.* 2013; Lovas and Torum 2001; Mork 1996; Pinsky *et al.* 2013), this contribution cannot be quantified as of yet, and so cannot feature in the analysis of the ecosystem service impacts of an MPA expansion scenario.

C3.4.3 Mangroves

As with seagrass beds, there exists a wide range of studies documenting that mangrove forests can attenuate waves (and sometimes substantially), and in so doing help to protect coastlines from erosive forces (Gedan *et al.* 2011; Thampanya *et al.*

³³ If this is not, in fact, problematic, then the relevant equations in Chen *et al.* (2007) are 12-15 on page 300.

2006; Tuyen and Hung 2010). Wave dissipation is strongly linked to the vegetation drag coefficients, and also depends on variables such as vegetation stiffness, vegetation height relative to water depth, initial wave heights, cross shore distances, mangrove forest structures (above and below the water and across space) and size (Alongi 2008; Gillis *et al.* 2014; Hu and Wroblewski 2009; Massel *et al.* 1999; Quartel *et al.* 2007; Tran Quang 2011; Tuyen and Hung 2010; Vo-Luong and Massel 2008). Also as was found in with seagrass, it is important to consider the effect of tidal currents (and tidal stage) on the ability of mangrove forests to attenuate waves (Alongi 2008; Hu and Wroblewski 2009; Luong *et al.* 2006).

The literature found also highlighted that the ability of mangrove forests to provide this service is not infinite. The capacity of mangrove forest to prevent erosion may be overwhelmed and subsequently undermined by large scale events such as regional erosion, river meandering, the decline of on-shore sedimentation, and large storm surges (Gedan *et al.* 2011; Winterwerp *et al.* 2013). It is also not the case that the provision of this service is guaranteed simply by the existence of mangrove forests. As (Tanaka 2009) shows, the spatial structure of mangrove forests can actually augment the impact of waves in certain instances by channelling the energy through a confined space. Ultimately, the provision of this service depends on the coincidence of the capacity, exposure, and the human demand for protection (Liquete *et al.* 2013). What this implies is that despite the fact that there is evidence that mangrove forests are nonlinear systems with nonlinear responses to changes (Gedan *et al.* 2011; Mazda *et al.* 2002), changes in the provision of this service cannot be estimated simply from considering changes in the extent of ecosystems unless one either considers (or assumes constant) exposure and human demand.³⁴

As with the searches conducted with respect to seagrass beds, although some larger scale reviews of mangrove forest-related wave attenuation were returned (Bostrom *et al.* 2011; Feagin *et al.* 2010; Gedan *et al.* 2011), and some studies were quite explicit in the mathematical modelling performed (Huang *et al.* 2011), these studies did not quantify changes in erosion rates or sedimentation as a consequence of wave attenuation. This means that this service cannot be considered further within the context of global MPA expansion scenarios.

C3.4.4 Coral reefs

There is some evidence that coral reefs can contribute to the provision of the erosion prevention service. Some research (based on the use of meters to measure current, tides, and waves in the field) found that the bottom friction coefficients associated with coral reefs are significantly (i.e. 10x) greater than that of sand or silt, and that the wave attenuation provided by coral reefs is positively correlated with the frequency of the waves travelling over the reef (Zhu *et al.* 2004). Other research has found that coral reefs can attenuate small amplitude tidal waves (Bouma *et al.* 2014), and that reefs can cause solitary waves to break further from shore, thus dissipating energy (Quiroga and Cheung 2013) and reducing the potential for erosion. The ability of coral reefs to attenuate waves (and therefore sediment transport and deposition patterns (Mandlier and Kench 2012)) may depend on coral cover (Villanoy *et al.* 2012) and the shape of reefs, as elliptical and circular reefs tend to retain more sediment, whereas sediment is more likely to be transported off reefs and beyond reefs when narrow and linear reefs are present (Mandlier and Kench 2012).

³⁴ A point that will have relevance in other biomes as well

Two studies³⁵ were found that explicitly modelled coastal sediment transport and/or erosion as a consequence of existence of reefs (Frihy *et al.* 2004; Lee *et al.* 2005). Lee *et al.* (2005) presents a numerical multi-module model “for predicting sediment transport and the associated erosion and deposition processes in a natural reef area (p. 303) that includes sub-models for predicting changes in wave heights, wave-induced currents, sediment transport (based on advection, dispersion, settling, and re-suspension), and coastal morphological changes. Within these sub-models, they also specify particular equations for seabed deposition and sea bed erosion. The former depends on “critical bed shear stress,” “the concentration [of suspended sediments] near the bottom,” and the “net sedimentation rate constant,” whereas the later depends on the “erodibility coefficient,” and “the critical bed shear required for re-suspension” (p. 304). However, these equations appear to only be usable as a part of the full model, the use of which is out with the scope of the analysis of global MPA expansion scenarios. Frihy *et al.* (2004) utilize both 1D and 2D simulation models to quantify the role that a fringing coral reef plays in the nearby beach erosion rates. The 2D model depends on the following variables: wave height, wave length, wave direction, and wave period, wave number, wave angle, on-offshore distance, longshore distance, wave energy, wave frequency, and wave group celerity. The outputs of these models are erosion estimates, measured in meters of coastline lost (i.e. an appropriate unit for this ecosystem service). However, the use of these models is also out with the scope of this project. Consequently, although there does appear to be some opportunities for gaining traction in quantifying the provision of this service, and how this might change with management, it remains a nontrivial task that appears to be most suited to localized case studies with access to the necessary resolution of oceanographic data.

C3.5 Waste treatment

C3.5.1 Seagrass

Seagrass beds can contribute to the provision of the waste treatment service by helping to bioremediate anthropogenic pollutants that are emitted into coastal waters. Various examples describing the provision of this service exist (e.g. Huesemann *et al.* 2009; Malea 1993; Malea *et al.* 1994; Marin-Guirao *et al.* 2005; Pennesi *et al.* 2013; Pennesi *et al.* 2012; Raghukumar *et al.* 2006; Solis *et al.* 2008). However, interpreting the existing literature in reference to this particular ecosystem service is difficult. For example, although hydrocarbons and polychlorinated biphenyls (PCBs) have been found to be degraded within seagrass beds, the evidence suggests that microbial communities, and not seagrass species, accomplish this degradation (Huesemann *et al.* 2009). It would appear, therefore, that seagrass species are essentially a part of the ecosystem structure that may then support the provision of the waste treatment service *by* microbial communities, but do not provide this service with respect to hydrocarbons and PCBs themselves.

³⁵ Barbier *et al.* (2011) was also considered for inclusion in this report, as this study presents equations that model wave height reductions as a function of distance from the edge of mangrove forests and marshlands, and as a function of water depth for seagrass beds and fringing coral reefs for particular case study locations around the world (Barbier *et al.* 2011). The study also considers wave height reductions in combination with variables such as water depth and area for mangroves (see Supplementary Information Barbier *et al.* 2011). However, wave height reduction is at best a proxy for the disturbance prevention and moderation service (Böhnke-Henrichs *et al.* 2013) – a service not considered within this project – and so this is not elaborated on more in this report.

Similarly, although the studies found highlight that seagrass species can be effective at biosorption of heavy metals (i.e. effective at removing these metals from the water column and sediments) (e.g. Pennesi *et al.* 2013; Pennesi *et al.* 2012), and so can be considered to be driving the partial remediation of the sediment and water column (as well as a bioindicator for water quality with respect to heavy metals (Gosselin *et al.* 2006; Lafabrie *et al.* 2007; Marin-Guirao *et al.* 2005)), this is not quite the same as indicating that they *remediate* these metal ions directly (i.e. subject them to reactions that result in less harmful ions). It is also unclear on what time scale this biosorption removes these pollutants from the wider environment and/or local food webs.

Additionally, it is important to note that literature found demonstrates that seagrass beds have a limited ability to bioremediate some of the common pollutants, such as sewage or nutrient-rich runoff, and are instead sensitive to said pollution. For example, the vitality of *Posidonia oceanica* exposed to sewage in Tunisia was found to have decreased substantially (as indicated by decreased leaf length, leaf surface area, the leaf area index, and the number and composition of seagrass epiphytes) as a consequence of said exposure (Mabrouk *et al.* 2013). Similarly, the shoot density of *Zostera noltii* was found to decrease with increasing concentrations of ammonia (Cabaco *et al.* 2008).³⁶ It should not be assumed, therefore, that the provision of the waste treatment service is inherently equivalent to the exposure of seagrass beds to anthropogenic pollutants, and instead efforts should be made to understand the capacity of seagrass beds to provide this service and how this capacity may vary with over-exposure to pollutants.

Ultimately, no study was found that quantified a clear relationship between the area or density or age of seagrass species and a capacity to bioremediate anthropogenic pollutants. When combined with some of the caveats in the literature discussed above, this highlights that in the absence of much more specific evidence, no assumptions can really be justified regarding the impact of expanding MPAs on the provision of the waste treatment service as mediated through changes in seagrass beds.

C3.5.2 Macroalgae

Macroalgae stands can contribute to the provision of the waste treatment service by helping to bioremediate anthropogenic pollutants that are emitted into coastal waters, including those emitted by aquaculture operations (e.g. Rodriguez and Montano 2007; Xu *et al.* 2008). Examples and highlights from the literature returned by the searches conducted to locate evidence related to the relationship between macroalgae and bioremediation are presented below (Table C6)³⁷. Although some evidence was found that macroalgae can remove heavy metals from the water column (Beolchini *et al.* 2009) (and potentially pass those metals up through food webs. See: Souza *et al.* (2012)), as well as support biofilms that are capable of remediating hydrocarbons (Radwan *et al.* 2002), much of the literature emphasized the uptake of nitrogen and phosphorus (and sometimes in the specific context of assessing the potential for macroalgae to function as biofilters in the context of integrated aquaculture production). These studies show that macroalgae can have extremely variable responses to exposure to nitrogen and phosphorus in various forms. This means that although there is the potential for expanded or healthier macroalgae stands to

³⁶ Note that decreasing shoot density may also affect the provision of the coastal erosion prevention service (see section C3.3.2)

³⁷ Note: Table 5 contains illustrative examples from the literature and is not exhaustive. In particular, many studies considering the uptake of nitrogen and phosphorus in various forms are omitted.

increase the provision of this service, the responses may be highly site specific and are not easily generalizable.

Table C8 Survey of the literature returned relevant to the bioremediation of pollutants by macroalgae stands

Species	State of Macroalgae	Pollutant(s) (place)	Key Points	Source
<i>P. palmate</i>	Live	Nitrogen (from the growth of 500 tonnes of farmed salmon)	1 ha of <i>P. palmate</i> could remove 12% of the waste N released	Sanderson <i>et al.</i> (2012)
<i>S. latissima</i>			1 ha of <i>S. latissima</i> could remove 5% of the waste N released	
<i>S. latissima</i>	Live	Nitrogen (from 5000 t salmon farm)	1 ha could remove: 0.36 t NH ₄ +-N (0.34% of dissolved inorganic N effluent in 11 months)	Broch <i>et al.</i> (2013)
<i>Palmaria palmata</i>	Live	Nitrogen species	0.49 mg N g ⁻¹ (of dry weight) day ⁻¹ (at 6 degrees C and 300 µM NO ₃ ⁻)	Corey <i>et al.</i> (2012)
<i>Chondrus crispus</i>	Live	Nitrogen	0.49 mgN gDW ⁻¹ day ⁻¹ (mean removal, independent of temperature and at 300 µM NO ₃ ⁻)	
<i>Gracilariaria verrucosa</i> (red algae)	Live	Phosphorus and Nitrogen	Maximum reduction efficiencies: PO ₄ -P: 58% NO ₂ -N: 48% NH ₄ -N: 61% NO ₃ -N: 47%	Huo <i>et al.</i> (2012)
<i>Pseudosciaena crocea</i>			Experimented with N and P removal potential in a lab-based setting.	
<i>L. japonica</i>	Live	Nitrogen and Phosphorus	In 36 hours of incubation, removed: N: 42-46% P: 35-45%	Xu <i>et al.</i> (2011)
			Note: this varied by temperature Measured removal of N and P in effluent from shrimp farm in Brazil.	
<i>Gracilaria caudate</i>	Live	Nitrogen and Phosphorus	Within 4 hours removed: NH ₄ : 59.5% NO ₃ : 49.6% PO ₄ : 12.3%	Marinho-Soriano <i>et al.</i> (2009)
			1 ha has potential to remove: N: 0.309 ton yr ⁻¹ P: 0.024 ton yr ⁻¹	

Table C8 Survey of the literature returned relevant to the bioremediation of pollutants by macroalgae stands (continued)

Species	State of Macroalgae	Pollutant(s) (place)	Key Points	Source
<i>Gracilaria birdiae</i>	Live	Nitrogen and Phosphorus	Measured removal of N and P in effluent from shrimp farm over 4 weeks. Removed: NH ₄ : 34% NO ₃ : 100% PO ₄ : 93.5% Considered in the context of fish farming	Marinho-Soriano <i>et al.</i> (2009)
<i>Gracilaria lemaneiformis</i>	Live	Nitrogen and Phosphorus	1 ha can remove: N: 0.22 t yr ⁻¹ from the water column P: 0.03 t yr ⁻¹ from the water column Within 3-4 days removed the following:	Zhou <i>et al.</i> (2006)
Porphyra species	Live	Nitrogen and inorganic Phosphorus	N: 70-100% (at concentrations up to 150 µM) P: 35-91%	Carmona <i>et al.</i> (2006)
<i>Ulva pertusa</i>				
<i>Gelidium amansii</i>	Live	Nitrogen	Experimented with N removal at different concentrations of NH ₄ and NO ₃ . Different species responded differently to the treatments. <i>U. pertusa</i> had the highest capacity for N removal (200 µMol/L)	Liu <i>et al.</i> (2004)
<i>Sargassum enerve</i>				
10 different types of macroalgae from the Arabian Gulf	Live	Hydrocarbons	This study found that macroalgae had biofilms that contained oil-utilizing bacteria that enabled the breakdown of hydrocarbons in the water carbon. These biofilms were not free-living and so depend on the macroalgae. Within 2 weeks these biofilms bioremediated the following: n-octadecane: 64-98% phenanthrene: 38-56% Measured sorption by macroalgae	Radwan <i>et al.</i> (2002)
Considered a variety of brown, green, and red algae	Live	Lead, Arsenic	Lead: Brown: 140 mg/g Green: 50-70 mg/g Red: 10-40 mg/g Arsenic: (at [Ar(V)] = 100 µg/L) Brown: □2 mg/g Green: □2 mg/g Red: □2 mg/g	Beolchini <i>et al.</i> (2009)

C3.5.3 Mangroves

Mangrove forests have the potential to contribute to the provision of this service in a number of different ways. Firstly, mangrove forests can remove heavy metals (e.g. mercury and methyl mercury, copper, zinc, lead, cadmium, nickel etc.) from the water column and concentrate them in either parts of the plant or facilitate their entrapment in sediments (Amat and Kassim 2010; Amusan and Adeniyi 2005; Bergamaschi *et al.* 2012; Che 1999; Machado *et al.* 2002; Naidoo *et al.* 2014; Nowrouzi *et al.* 2012). Similarly, mangroves can function as trace metal sinks (Suzuki *et al.* 2014), and under certain conditions, mangroves also sometimes experience the formation of an iron plaque on their roots that immobilizes heavy metals (Pi *et al.* 2011). Secondly, mangrove forests can support microbial and fungal populations capable of degrading hydrocarbon pollutants (Guo *et al.* 2012; Ke *et al.* 2003; Ruiz-Marin *et al.* 2013; Santos *et al.* 2014; Wang *et al.* 2014; Wongwongsee *et al.* 2013; Wu *et al.* 2010), though the capacity of these populations to bioremediate hydrocarbons depends on the specific microorganisms found in any given location, the exposure to waste hydrocarbons, and certain features of the water column (e.g. nutrient concentration, salinity, temperature) (Santos *et al.* 2011).

Thirdly, mangroves can have nontrivial uptake capacities with respect to nitrogen and phosphorus enrichment (Lambs *et al.* 2011), and have been used as biofilters to partially³⁸ remediate agricultural, human, and aquaculture effluents through the uptake of nitrogen and phosphorus into plant tissues (Chen *et al.* 2011; Huang *et al.* 2012; Moroyoqui-Rojo *et al.* 2012; Zaldivar-Jimenez *et al.* 2012). This uptake of additional nutrients has been found in Thailand to correlate with the diversity of select key species, indicating the conservation objectives may be compatible with the provision of this service (Wickramasinghe *et al.* 2009).

That said, mangrove forests are also capable of releasing heavy metals, excreting them³⁹, or failing to sequester them (Bergamaschi *et al.* 2012; Naidoo *et al.* 2014), depending in part on plant age and biomass production and other local environmental variables like salinity (Chang *et al.* 2009; Tam and Wong 1997). Furthermore, it should be noted that there is a potentially nontrivial trade-off between the uptake/retention of heavy metals by mangroves and the productivity, health, and stability of mangrove forests (Cheng *et al.* 2012; Huang and Wang 2010; Khan *et al.* 2013; Naidoo *et al.* 2014), and by extension possibly other ecosystem services that depend on the health of mangrove forests. Finally, it is worth noting two points regarding nutrient enrichment and mangrove forests: 1) nutrient enrichment from effluent tends to involve increased sedimentation, and there are limits to the rate of sedimentation that mangroves can withstand before dying (Vaiphasa *et al.* 2007); 2) nutrient enrichment has the potential to alter the dynamics of carbon sequestration within mangrove forests such that mangroves end up venting CO₂, N₂O, and CH₄ that would otherwise have not been vented from the mangroves (Chen *et al.* 2011; Suarez-Abelenda *et al.* 2014). Thus, there may be a trade-off in certain circumstances between the provision of the waste treatment service and the climate regulation service.

³⁸ 'Partially' this context may mean the overwhelming majority (i.e. up to 88%) of N or P within the effluent (Huang *et al.* 2012)

³⁹ For example, Naidoo *et al.* (2014) found that copper and zinc were excreted through mangrove leaves, making these metals available to enter the surrounding environment again, whereas lead and mercury were not excreted through the leaves. The accumulation of heavy is species-specific as is the ultimate storage location of the metals once taken up by the plant (Akhand *et al.* 2012).

Overall, therefore, although there is clear evidence that mangroves do contribute to the provision of the waste treatment service, because of the caveats highlighted above and because no studies were found estimating unit area capacities for remediation, this service cannot be considered further in the context of a global MPA expansion scenario.

C3.5.4 Coral reefs

None found.

C3.6 Lifecycle maintenance

In order to assess the provision of lifecycle maintenance, it is necessary to consider definitions of marine nurseries in the context of an ecosystem services lens.

Nagelkerken (2009) define nurseries as follows: "Habitats are considered nurseries if their contribution, in terms of production, to the adult population is greater than the average production of all juvenile habitats, measured by the factors density, growth, survival, and/or movement" (p.357). Similarly, Dahlgren *et al.* (2006) defines them as "...a marine nursery is defined as a juvenile habitat for a particular species that contributes a greater than average number of individuals to the adult population on a per-unit-area basis, as compared to other habitats used by juveniles (p. 291). Sheridan and Hays (2003) consider a nursery to be "...a special place for juvenile nekton (fishes and decapod crustaceans) where density, survival, and growth of juveniles and movement to adult habitat are enhanced over those in adjoining juvenile habitat types." (p. 449). Essentially, nurseries are areas of increased juvenile survival (Grol *et al.* 2011) that export non-juveniles to different habitats. The ES typology employed for this study further focuses the lifecycle maintenance service on those species that *use* marine nurseries that are later of commercial importance out with the nurseries. Commercial importance can stem from either harvesting (in the case of fisheries) or tourism. Thus, when considering the provision of the lifecycle maintenance service, quantitative evidence is needed regarding the *production* by nursery areas of species that are of direct commercial importance.

Many studies content that mangrove forests and seagrass beds are marine nurseries for a variety of species, including reef fish and reef sharks (e.g. Chin *et al.* 2013; Nagelkerken 2009), though through the early 2000s contentions that sites and habitats were nurseries was not frequently supported by sufficient quantitative sampling (see Sheridan and Hays 2003). More recent research suggests 1) that in at least some instances the size and connectivity of estuary habitats (combinations of mangrove forests, salt marsh, and seagrass beds) correlates significantly with fish catch outside the estuaries (Meynecke *et al.* 2007), but also 2) that the use of particular habitats (or habitat types) as nurseries is highly variable, that nursery use is species-specific. By extension, relationships and trends regarding nursery value and use cannot be generalized at the family level (Jaxion-Harm *et al.* 2012) and *should not* be generalized for any particular site *a priori*, as habitat configuration and connectivity may be more important than habitat type (Dorenbosch *et al.* 2007).

Considering nurseries in the context of MPAs is additionally complicated because of the effects that MPAs can have on trophic interactions are also relevant to the ability of a particular habitat to function as a nursery for a particular species. As Planes *et al.* (2000) discuss in their analysis of the effects of MPA designation on fisheries recruitment in Mediterranean case studies, the size, location, and condition of the MPA can undermine the ability of nurseries to support recruitment (e.g. if an MPA enables

the population of predators to recover). Consequently, this section will not seek a global, generalized relationship on the provision of the lifecycle maintenance service. Instead, the rest of this section will explore the evidence found pertaining to the ability of individual habitats to function as nurseries and the conditions under which they might so function, as well as highlighting individual case studies where quantitative information was found.

C3.7.1 Seagrass

Seagrass beds can contribute to the provision of the lifecycle maintenance service by providing habitats for the juvenile life stages of species that are of commercial importance and that are harvested (or observed or collected) in a different habitat. The literature returned by the searches conducted to locate evidence related to the relationship between seagrass and lifecycle maintenance revealed a wide range of evidence supporting the idea that seagrass beds do function as nursery areas for commercially important species. For example, Warren *et al.* (2010) presents evidence that juvenile cod density responds annually to changing eelgrass cover in the case of both Atlantic and Greenland cod, and Joseph *et al.* (2006) found that eelgrass as the sole nursery in eastern Canada for white hake (*Urophycis tenuis*) and (small, <3cm) cunners (*Tautoglabrus adspersus*). Similarly, Polte and Asmus (2006) found that *Zostera noltii* beds were spawning grounds for *Belone belone*. (Verweij *et al.* 2008) found that nearly 98% of juvenile yellowtail snapper fish (*Ocyurus chrysurus*) spent time in seagrass meadows as juveniles.

Additionally, within the literature focused on investigating the extent to which seagrass beds can function as nurseries, there are some studies that explicitly consider seagrass beds within MPAs. Bussotti and Guidetti (2011) considered 22 taxa of juvenile fish and 10 different habitat types across a full calendar year within the Torre Guaceto MPA in the southeast Adriatic Sea. They found that *Posidonia oceanica* beds were home to several species (*Chormis chormis*, *Spondiosomo cantharus*, *Diplodus annularis*, and *Dicentrarchus labrax*), and suggest that by protecting seagrass beds, the MPA can help to sustain the local fish diversity.⁴⁰

A range of studies also focused on the role that seagrass beds play in supporting ontogenic migrations from seagrass beds (and also mangroves) to coral reefs (Berkstrom, Jorgensen, *et al.* 2013; Berkstrom, Lindborg, *et al.* 2013), though this effect was not found to be universal (e.g. Nakamura and Sano 2004). Campbell *et al.* (2011), for example, focused on ontogenic migrations with an Indonesian MPA and found that there were different species, life stages, and feeding groups located along the transition from seagrass bed to coral reef, supporting the notion that seagrass beds can provide the lifecycle maintenance service for reef fish. Importantly, however, this study highlighted that the details of the results found (in terms of which species were found where, and during what life stages) do differ between studies. This is also supported by Huijbers *et al.* (2008), a study that found that some reef species are flexible in terms of the habitats they can use as juveniles, by Chittaro *et al.* (2005), a study that found there was only limited connectivity between certain shallow reef

⁴⁰ As an aside, it is interesting to note that this study did not consider the question of whether or not the MPA in question was exclusively a sink of fish larvae rather than also a source of adults. As highlighted in section 3.1.3, some existing MPA modelling-based studies indicate that if MPAs are located exclusively at sink sites may actually undermine the sustainability of fisheries. Applied to this case, it means that if this MPA protects nursery areas for the species highlighted, but not the adult habitats as well, the MPA may be undermining the realization of the commercial impact of protected nurseries despite protecting the juvenile populations.

systems and local mangrove stands and seagrass beds within the Caribbean, and by Nakamura (2010), a study that found that some fish species around Ishigaki Island (Japan) either declined dramatically or disappeared following the destruction of the seagrass habitats in a typhoon. The implication of this is that one cannot assume that a particular species will utilize seagrass beds as a nursery in a certain area without field data to support that assumption. By extension, this means that one cannot assume that the provision of the lifecycle maintenance service is automatically increased by the protection and/or expansion of seagrass beds.

Furthermore, the collection of studies returned highlighted the importance of recognizing potential edge effects created by the distribution of seagrass beds and their relative patchiness in relation to the provision of this service. Carroll and Peterson (2013), for example, compares scallop survival and growth rates in seagrass beds, out with (but near) seagrass beds, and on the boundaries of seagrass beds. They found that although scallop survival was greatest within seagrass beds,⁴¹ scallop growth rates were lowest there. In contrast, scallop survival was lowest on sandy environments, but their growth rates were the greatest there. The edge of seagrass beds provided intermediate survival and growth rates.

Edge effects were also recently considered within Philipa Bay, Australia in relation to fish assemblages and both shallow water (<1.5 m) and deep water (3.5-6 m) seagrass beds (Smith *et al.* 2012). This study found that different species tended to inhabit different depths, and also that longer species tended to inhabit the edge of the seagrass beds rather than the middle of the beds. These studies highlight that there may be trade-offs between protection and growth for species that do utilize seagrass beds, and also that the spatial distribution of seagrass beds may be important to consider in addition to total seagrass extent.

This theme of the importance of the role that habitat structure plays in the achievement of certain ecological outcomes also emerged from a review of more than 200 studies that were relevant to the hypothesis that seagrass beds function as marine nurseries (Heck *et al.* 2003). This study was restricted to studies that made some type of comparison between seagrass beds and other habitats with respect to the density, growth, survival, and migration of the targeted species. The results of the review indicated that there was data that seagrass beds supported higher abundance, growth, and survival rates than did unstructured habitats (and that this effect was potentially more important in the northern hemisphere than the southern hemisphere). Importantly, however, the review did not find substantive differences between seagrass beds and *other structured habitats* (e.g. oyster reefs, macro algae stands, mangrove forests). This review also did not find evidence of commercial harvests decreasing in response to declining seagrass beds (though other published studies do suggest this is a potential outcome of declining seagrass beds (Halliday 1995; Heck *et al.* 1995; McArthur and Boland 2006).

Similarly, de la Moriniere *et al.* (2002) compared mangrove forests, seagrass beds, and coral reefs with regards to their populations of juveniles for 9 different reef fish species in the Netherland Antilles. They found that were as some species utilized only one of these habitats as juveniles, others used a mix of different habitats. They further

⁴¹ Interestingly, survivorship within a seagrass bed may change for some species with increasing size (either in absolute terms or relative to unvegetated neighbouring habitats). There is some evidence that this is the case with Caribbean spiny lobster (Lipcius *et al.* 1998) and blue crabs within the Chesapeake Bay (Pile *et al.* 1996).

identified three models for the post-settlement life cycle migrations: long distance migrations (e.g. from mangrove forests or seagrass beds to a reef), short distance migrations (e.g. where settlement is in close proximity to reefs or on the reef), and step-wise migrations (e.g. where multiple habitats are utilized in different stages as the individual matures and moves progressively closer to the reef). de la Moriniere *et al.* (2002) also contend, as do Pollux *et al.* (2007), that site selection for larval settlement may be at least somewhat active, rather than purely stochastic. All of this suggests that there is a need to identify those circumstances where a specific species (e.g. *Posidonia oceanica*), as opposed to a generically structured habitat (or the coincidence of structured habitat with hydrographic features (see: Stoner 2003), is necessary for the provision of the lifecycle maintenance service.

Finally, still other research indicates there is a need to identify those circumstances where a structured habitat is important at all. Jackson *et al.* (2002), for example, compared those species that were associated with a *Zostera* bed and those species that were associated with sandy flats across different parts of the tidal cycle and did *not* find any evidence to suggest that the *Zostera* beds supported higher densities of commercially valuable species than did the sand flats. Similarly, Schaffmeister *et al.* (2006) found that some shrimp species (e.g. *Penaeus kerathurus* and *Penaeus notialis*) will utilize *both* tidal flats and seagrass beds prior to migrating offshore as adults.

Despite research such as that cited in the preceding text, only one study – McArthur and Boland (2006) - was found that explicitly focused on quantitatively estimating the relationships between changing seagrass area and some other metrics that may signal the provision of this service such as changing adult biomass or indeed actual harvests (Table C7⁴²). This is as opposed to trying to document juvenile abundance within potential nursery areas (e.g. Bertelli and Unsworth 2014) or monitor juvenile growth within seagrass beds (e.g. Jones 2014). This may be because even the form of this relationship is unclear (i.e. additive, multiplicative, etc.) (McArthur *et al.* 2003). The relationship utilized within this study is generic in its form, but site-specific in its parameterization. Therefore, if there is sufficient data to justify the assumption that particular areas of relevance to the global MPA expansion scenario are, in fact, nursery areas, then at least the generalized catch equation from McArthur and Boland (2006) (see foot note 42) could be used. Its application, however, would require the estimation of new parameters, including values for the seagrass residency index (SRI) for each species in each area under consideration. Such a task may be beyond the scope of this study, despite its potential.

⁴² Note: Table 6 contains illustrative examples from the literature and is not exhaustive.

Table C9 Summary of studies relevant to understanding the provision of the lifecycle maintenance service by seagrass beds

Seagrass species	Country	Species	Summary	Key outputs	Source
Un-specified (implied mix of species)	Australia (south)	58 species, evaluated based on residency within seagrass	This study uses models to link seagrass area to secondary fish production outside these seagrass beds and ultimately estimate a value impact per unit area	Catch-Seagrass-Effort models⁴³ All the models estimated were significant at the 0.05 level and featured R^2 values between 60% and 97%. The catch estimates were further decomposed into commercial, recreational, and discard by using the following estimated relationships: $C_{total,i} = C_{com,i} + C_{rec,i} + C_{dis,i}$ $C_{rec,i} = 0.25C_{com,i}$ $C_{dis,i} = 0.286C_{com,i}$ Assumptions: Catch in linear in effort, but the parameter in this equation is a function of seagrass area. One of the parameters of this sub-function is the seagrass residency index (SRI) to enable the distinction between species that spend a lot or a little time within seagrass	McArthur and Boland (2006)

C3.7.2 Macroalgae

One study was found that was relevant to this service. The study identified focused on the Wadden Sea found that *Fucus vesiculosus* stands contained 20x the number of herring eggs than did other habitats (Polte and Asmus 2006), indicating macroalgae stands can, in at least certain circumstances, function as a marine nursery for commercially important species harvested elsewhere. None of the evidence found, however, was sufficient to support the analysis of changes in the provision of these ecosystem services in response to MPA designation.

C3.7.3 Mangroves

Prior to 2003, mangrove forests had been hypothesized and assumed to be marine nurseries (and by extension, providers of the lifecycle maintenance service), but the support for this hypothesis in the literature was undermined by the following features of existing studies: studies often utilized inadequate (and sometimes confounding) approaches to sampling, few studies made explicit comparisons to other habitats, and there was insufficient quantitative data available to assess the effects of sheltering in mangroves on growth or survival of individuals, or on adult population sizes (Clynnick and Chapman 2002; Halpern 2004; Sheridan and Hays 2003).

There appears to be more recent evidence, however, to more robustly support the notion that mangroves can function as marine nurseries for commercially important species (and therefore can provide the lifecycle maintenance service) (Nagelkerken *et al.* 2002; Nagelkerken and van der Velde 2002a, c). Mangroves appear to play a particularly important role in supporting coral reef biomass, and many coral reef species (at least within the Caribbean) appear to have an “obligate dependence” on mangrove forests during their juvenile life stages (Nagelkerken 2007). This

⁴³ The generic model form used is as follows: $C_i = e^{\beta_{1i}}(s + \lambda_i)^{\beta_{2i}} + \varepsilon_i$ where λ_i is the median area of seagrass in the grid where the fish is targeted, β_{2i} is the SRI value for a given species, s is seagrass area, and β_{1i} is obtained using least squares.

dependence results in ontogenic migrations from mangrove forests to coral reefs as individuals mature. A large number of studies (utilizing diverse methods such as stable isotope analyses, fish gradient construction, and statistical population tracking) have documented this mangrove forest-coral reef connectivity and sometimes across nontrivial distances (e.g. tens of kilometres) (Aburto-Oropeza *et al.* 2009; Jones *et al.* 2010; Kimirei *et al.* 2013; McMahon *et al.* 2012; Nagelkerken and van der Velde 2002a; Vaslet *et al.* 2010).

Furthermore, there is some evidence that mangroves can, depending on their degree of connectivity with wider ecosystems, support ecological parameters such as reef biomass (Mumby 2006) and anthropogenic parameters such as offshore fishery yields. Chong (2007), for example, contends that 556,856 ha of mangrove forest in Malaysia support more than 50% of the 1.28 million tonnes of annual offshore fishery landings. There is even evidence that replanted mangroves can act as marine nurseries (Crona and Ronnback 2007), indicating that there may be the potential to recover provision of the lifecycle maintenance service that has been lost as a consequence of past clearances of mangroves forests.

It is important to note, however, that not all mangroves function as nurseries,⁴⁴ and that the use of mangroves is highly variable at the species level. Some species appear to be solely dependent on mangrove forests during their juvenile life stages (Laegdsgaard and Johnson 1995), whereas evidence from stable isotope, amino acid, and visual survey analyses demonstrate that other species (and in some locations most coral reef species Olds *et al.* 2012; Unsworth *et al.* 2009) utilize a wide range of habitats (in some instances for different purposes within a single life stage and in other instances for different phases of development) prior to reaching full maturation (Kimirei *et al.* 2013; McMahon *et al.* 2011; Nyunja *et al.* 2009). For example, the French grunt has been found to rely on mangrove forests for one life stage and seagrass beds for another life stage (Grol *et al.* 2014), and it is estimate that 20% of commercially important fish species considered in a recent study in the Philippines rely on multiple habitats as juveniles (Honda *et al.* 2013). Additionally, in environments where tidal fluctuations fully expose mangroves, species may need to routinely utilize an alternative habitat, such as seagrass beds (Jelbart *et al.* 2007; Sheaves 2005). In areas featuring connectivity between coastal mangrove forests and coral reefs, the extent to which a mangrove forest is utilized as a nursery may also depend on the distance that the target coral reef is from shore to which the juvenile fish will later need to migrate. McMahon *et al.* (2012), for example, documented that near shore habitats (such as mangroves) were much more frequently utilized as nurseries by *Lutjanus ehrenbergii* that targeted nearshore coral reefs, whereas oceanic reefs became much more important when the end destination was a reef further 30-50 km offshore.

The implications for the analysis of the provision of the lifecycle maintenance are twofold. Firstly, this implies that in at least some instances the service is actually provided by a location-specific suite of habitats, rather than a single habitat. This is further supported by Kopp *et al.* (2010), a study that found that fish assemblages found within a nursery habitat (i.e. seagrass beds) depend on what the adjacent habitats are, and by Unsworth *et al.* (2008) that concluded there needed to be explicit recognition of the fact that multiple habitats interact provide marine nurseries. Given this, it would be best (albeit not possible given the available data) to try and include

⁴⁴ Barnes *et al.* (2012) found that estuarine and clearwater mangroves in the IndoPacific do not appear to be marine nurseries. Lee *et al.* (2014) also suggests this is not a ubiquitous function of mangrove forests

recognition of this within the analysis of scenarios that feature the protection and/or improvement and/or expansion of habitats like mangrove forests and seagrass beds.

Secondly, it implies that the geospatial positioning of the MPAs relative to underlying marine habitats may be more important in supporting marine species assemblages than is simple extent of a particular habitat (e.g. a coral reef) that is included within an MPA (Olds *et al.* 2012). By extension, simple estimates of increased area protected (or recovered), or even the simple presence/absence of mangroves (Nip and Wong 2010), may not be reasonable proxies for changes in the provision of this service in the absence of other data that documents how these habitats are used locally in conjunction with other habitats by different species of interest. Further support for this can be found in Faunce and Serafy (2008), a study that argues that even across a given shoreline not all mangroves are equivalent in terms of their ability to be nurseries, concluding that simple assessments of total habitat area will “grossly overestimate” the extent of true nursery habitat in any given area. Similarly, Drew and Eggleston (2008) highlights that there are species-specific scale effects related to nursery use that can only be investigated through research efforts such as individual-based modelling and landscape-scale analyses.

If, for the purposes of the analysis of a global MPA expansion scenario, there is a desire to assume that simple changes in area correspond in a straight forward way to changes in the provision of a marine nursery (and the carrying capacity of that nursery), and therefore to changes in fishery yields out with the mangroves, the approach illustrated in Barbier and Strand (1998) has potential. In order to do this it must be assumed that there is a stock X , measured in biomass units and that the stock size changes in time as a consequence of biological logistic growth. It must also be assumed that harvesting “follows the Schaefer production process” (p. 58). This yields the following dynamic relationship between long run equilibrium catch and mangrove area (Eq. C11):

$$h = q\alpha EM - \frac{q^2}{r} E^2 \quad (\text{Eq. C11})$$

Where h is catch, q is a catchability coefficient, α is the constant in the relationship between carrying capacity (K) and mangrove area (M) (i.e. $K=\alpha M$), E is fishing effort, and r is the intrinsic species growth rate. The parameters $q\alpha$ and q^2/r can be estimated using time series data of harvests and mangrove area.

Another possibility is to draw on the equations collected by Manson *et al.* (2005). These relationships relate to prawn and fish production from mangroves in parts of Indonesia, the Philippines, Malaysia, Australia, the Gulf of Mexico, and Vietnam. A global tropical prawn production equation, and a hemispheric-scale prawn production equation are also featured in Manson *et al.* (2005). As this shows, there are global, generic equations related to prawn catch and changes in area of “intertidal vegetation,” as well as region-specific relationships between fish catch and mangrove area. Depending on the specific nature of the global MPA expansion scenario, one or more of these relationships may be more easily utilized than the Barbier and Strand (1998) equation shown above.

Overall, therefore, there are equations that can potentially be utilized to analyse changes in the provision of the lifecycle maintenance service, if it is assumed that MPA implementation also results in changes in the area of mangrove forests, and that simple changes in area can be taken as a proxy for changes in the carrying capacity of the relevant nursery areas. It must also be assumed that the MPA does not also encompass the full home range of the species utilizing the mangroves as a nursery

area, as this would preclude the provision of this service. Finally, it is worth noting, that there may be other attributes of mangrove forests that affect the provision of this service, none of which are reflected in the relationships shown in this section (Manson *et al.* 2005).

C3.7.4 Coral reefs

Although there is evidence that coral reefs are connected to marine nurseries, there is little evidence that coral reefs are themselves nurseries (i.e. that they host the juvenile life stages of commercially important species that are harvested/extracted elsewhere). That said, Foley *et al.* (2010) employ the approach demonstrated in Barbier and Strand (1998) to quantify relationships between redfish harvest and cold water coral reefs in Norway. If relevant to the global MPA scenario analysis, the relationships estimated in Foley *et al.* (2010) could potentially be used to provide ballpark estimates of changes in the provision of the lifecycle maintenance service over a wider geographic scale than Norway.

C3.7 Recreation & tourism

C3.8.1 Seagrass

Although only a small pool of literature was found related to seagrass beds and tourism, it is clear from the studies found that seagrass beds can have positive or negative impacts on recreation and tourism. They can contribute to the provision of recreation/tourism in that they can be habitats that are attractive for diving, snorkelling, and recreational fishing (Vlachopoulou *et al.* 2013), but at least in the case of *Posidonia oceanica* in the Mediterranean Sea they can also undermine recreational experiences when they are deposited (and subsequently decompose) on public beaches (De Falco *et al.* 2008). Daby (2003) also documented some (largely unfounded) concerns in Mauritius by some hotels that swimmers would find seagrass to be unsightly and/or that it would hide marine species that were a threat to safe swimming.

It is also clear, however, that recreational activities can threaten seagrass beds. Recreational boating can scar seagrass beds and the species that reside within the beds (Bishop 2008; Burfeind and Stunz 2006; Burfeind and Stunz 2007), as can anchoring (Hallac *et al.* 2012; Okudan *et al.* 2011). Because some species of seagrass (such as *Posidonia oceanica*) are slow to recover from human disturbance and damage (Boudouresque *et al.* 2009), there is a need to try and ensure that recreation that is pursued in the vicinity of seagrass beds does not damage those beds. The implication for MPA management is that in addition to excluding commercial activities, the sustainability of recreational activities needs to be actively managed, and with respect not just to charismatic species, but also the habitats found within the MPAs.

C3.8.2 Macroalgae

In terms of tourism, there were several studies that contended that increased macroalgae production undermine beach-based tourism (Charlier *et al.* 2008; Morand and Briand 1996; Smetacek and Zingone 2013).

C3.8.3 Mangroves

The literature searches conducted returned little information on the relationship between mangroves and tourism/recreation. There is some evidence that ecotourism visits to mangroves do happen (Avau *et al.* 2011), and that the installation of infrastructure (such as boardwalks) can increase the potential for mangrove forests to support recreation and education (albeit with increased environmental damage) (Kelaher *et al.* 1998). There is also a need, however, to better understand the pressures that would be exerted on mangroves by increased tourism (Kelaher *et al.* 1998). No quantitative data was found on tourism in the context of mangroves, and so this service cannot be taken further in the analysis of global MPA expansion scenarios.

C3.8.4 Coral reefs

It is clearly the case that coral reefs are an important destination for tourism (see for example: Hasler and Ott 2008), and that they can provide ecological support to tourist activities (see for example Henry *et al.* 2013; Ruiz-Frau *et al.* 2013). Some researchers have suggested that improvements in the health of coral reefs (and by extension marine biodiversity) may improve the value of reef-related tourism (Schuhmann *et al.* 2013; Williams and Polunin 2000), where declines in coral health (and marine biodiversity) may result in the decline of marine tourism (Kragt *et al.* 2009). Other recent research has documented the existence of positive economic and educational impacts in local communities in response to MPA-related tourism (Daldeniz and Hampton 2013).⁴⁵

That said, insufficient evidence was returned from the literature searches to understand how tourism and recreation changes in response to MPA designation, improvements or declines in coral reef health, or changes in coral reef extent. For example Dicken (2014) documented that 59,553 dives were conducted by 15,780 divers in the St. Lucia and Maputaland MPA in South Africa, that 95.2% of these dives occurred on coral covered sandstone reefs, and that 84.2% of respondents were interested in opportunities for pursuing shark diving. This data hints at there being a potential role for the MPA to play in increasing tourism, as the MPA can help to protect those features (i.e. coral-covered sand stone reefs and sharks) that attract divers. However, without baseline data, data regarding how tourism numbers have changed with time, and an analysis of confounding variables, it is not possible to quantify the impact that the MPA designation had on this tourism. Similarly, Ahmad and Hanley (2009) document that the number of visitors to Payar Marine Park increased 3,668-133,775, but do not focus in any detail on why tourism has increased, focusing instead on the results of a non-market valuation study conducted.

The lack of attention to the drivers of tourism change and the lack of analysis of potentially confounding variables (i.e. contextual variables) may be especially important in the context of the development of tourism as a viable form of alternative livelihood in developing countries. As argued by Wood *et al.* (2013), each of the following must be in place before a catch and release sport fishing sector (that could benefit from MPA designation) would be viable in a developing country context: local capacity to manage tourism and tourist facilities must exist and be supported by co-management of stakeholders across different scales of activity; equitable benefit sharing arrangements should be in place and backed by government; resource

⁴⁵ Note that these benefits were also paired in this case study with the commodification of cultural traditions (i.e. a negative socio-cultural impact). This specifically occurred in response to dive tourism on the 3 Malaysian islands of (Perhentian, Redang, and Mabul).

boundaries and rights must be clearly delineated; clear pathways to impact on health, education, food security, and species biomass must have been found; monitoring and evaluation processes and procedures must be agreed to and in place. What this means is that simply designating an MPA (or expanding an MPA) may not be sufficient to realize potential (beneficial) increases in the provision of tourism and recreation.

In contrast to the lack of quantitative evidence documenting how tourism changes in response to MPA designation, evidence was found to that suggests, at least in the case of the Great Barrier Reef, that visitor numbers and frequency depend on a set of complex relationships between environmental, operational, and customer service attributes, rather than just on environmental attributes (Coghlan 2012). Other research focused on the Great Barrier Reef identified a series of “meaning themes” that provide insights into what attracts people to the reef (Wynveen *et al.* 2010). As with Coghlan (2012), the layers of meaning ascribed to the Great Barrier Reef were not all environmental, strictly speaking.⁴⁶ Similarly, other research has shown that different types of tourists have different environmental preferences, meaning that there can be conflict and tension between different types of boaters and other activities such as whale watching (Gray *et al.* 2010), or that tourism can grow without reference (or sensitivity to) environmental health (Carr and Heyman 2009). Because MPA designation and enforcement targets only environmental attributes, the implication if visitor numbers and frequency are similarly affected elsewhere is that tourism impacts may not be inferable simply from consideration of MPA features.

It may also be the case that in at least some instances promoting tourism as a (economically and environmentally) sustainable activity may be counterproductive to the conservation of coral reefs. For example, recent research indicates that tourism rates explained 84% of the variability in the $\delta N-15$ signatures found in sea fans in Quintana Roo, Mexico (Baker *et al.* 2013). This study highlights that the presence of tourism can result in increases in pollution that undermine the species that conservation measures like MPAs are intended to protect. Hassanali (2013) considered the Tobago Bucco Reef Marine Park and also found there is some (albeit unquantified) relationship between increasing tourism and the decline of the coral reefs within the MPA. Tourism has also been linked to coral disease occurrence (Lamb and Willis 2011). The mechanism at work here may have something to do with sunscreen (Danovaro *et al.* 2008), but the details of this link between tourism and the facilitation of coral disease are still quite unclear (Lamb and Willis 2011).

Tourism (largely in the form of SCUBA diving and fishing) has been similarly implicated as one of the causes of coral reef decline elsewhere in the China Sea, the Great Barrier Reef, the Mediterranean Sea, the waters off of eastern South Africa and Mozambique (Brodie and Waterhouse 2012; Currie *et al.* 2012; Linares *et al.* 2012; Zhao *et al.* 2012). As mentioned in section C3.2.2, the act of participating in tourist activities within marine environments (like SCUBA diving) may damage coral reefs (Chung, Au, *et al.* 2013). It may be possible to reduce or control (although not fully eliminate (Leujak and Ormond 2008)) these impacts through the specification of codes of contact, educating especially inexperienced divers, designating underwater trails, more carefully considering access points, using tourist carrying capacities to limit visitation numbers, improved environmental planning, increased awareness to cultural context, and monitoring human impacts more closely (Anderson and Loomis 2011; Hunt *et al.*

⁴⁶ The themes identified were: “Aesthetic Beauty, Lack of build infrastructure/pristine environment, Abundance/diversity of coral and other wildlife, Unique natural resource, Facilitation of desired recreation activity, Safety and accessibility, Curiosity and exploration, Some connection to natural world, Escape from every day, Experience with family and friends” (Wynveen *et al.* 2010).

2013; Meyer and Holland 2008; Ong and Musa 2011; Rios-Jara *et al.* 2013; White *et al.* 1997), and produce a more sustainable form of tourism within MPAs (e.g. Hawkins *et al.* 2005). However, the available literature indicates there is a need to explicitly consider tourism (and tourist behaviour) to be a pressure on, rather than just an ecosystem service provided by, coral reefs, and to use the tools available such as Monte Carlo-based forecasting models of Saphier and Hoffmann (2005) to try and anticipate and pre-empt damage to coral reefs from recreational activities.

What the aforementioned means in the context of this study is that although it can be assumed that a global expansion of MPAs should impact on tourism *somewhere*, it will not be possible to quantitatively estimate the magnitude of this impact, to locate it spatially. That said, as a consequence of the spatially-explicit meta-analysis presented in Ghermandi and Nunes (2013) it should be possible to identify whether proposed coastal MPAs are located in proximity to coastal areas with a high recreational value.

C3.8 Air purification

C3.8.1 Seagrass

None found.

C3.8.2 Macroalgae

None found.

C3.8.3 Mangroves

The literature search conducted that was intended to identify extant evidence related to the provision of the air purification service by mangrove forests returned one study the commented directly on this service: Naidoo and Chirkoot (2004). This study found that the leaves in a mangrove forest downwind of a coal emissions do remove the coal dust from the atmosphere (and so provide the air purification service). However, this study also notes that the presence of coal dust on the mangrove leaves reduces CO₂ exchange in *Avicennia marina* by 17-39%. This implies there may be trade-offs between the provision of the air purification service and other ecosystem services such as the climate regulation service.

C3.8.4 Coral reefs

None found.

C3.9 Cultural heritage and identity

C3.9.1 Seagrass

There appears to be very limited evidence regarding the role that seagrass has in the direct provision of the cultural heritage and identity service. Only a single study was found that addressed this - Turner (2001) - and this study documents that both algae and seagrass appear within the narratives and traditions of the First Peoples on the northwest coast of North America. Based on this limited pool of information it will not be possible to assume or suggest anything regarding changes in the provision of the

cultural heritage and identity service following the expansion and/or recovery of seagrass beds as a consequence of MPA designations.

C3.9.2 Macroalgae

In terms of cultural heritage and identity, the only evidence found came from Chile (Vasquez *et al.* 2014). This study conducted used contingent valuation to elicit an existence valuation that the authors argued referred to a mix of ecosystem services, one of which could be cultural heritage. The treatment of cultural heritage was extremely vague in this study, and ultimately the research found that the economic value of kelp as a source for alginate was much more significant than the value associated with cultural heritage and identity and did not consider cultural heritage to be an important service provided by kelp stands.

C3.9.3 Mangroves

Only one study was found that attempted to address the connection between mangroves and cultural heritage and identity: James *et al.* (2013). This study documented the percentage of respondents in three locations who responded “yes” to questions asking if they felt that mangroves in the Niger delta provided things such as “therapeutic value,” “amenity value,” “heritage value,” “spiritual value,” and “existence value.” However, this study was not well contextualized with respect to the existing ecosystem services literature, and did not appear to include any effort to understand how these answers connected to the health and/or state of mangrove forests. Consequently, this study cannot be used as a basis for analysing changes in the provision of this service in the context of this project.

C3.9.4 Coral reefs

Two studies were found that discussed the cultural dimension of coral reefs: Hicks *et al.* (2013); Moberg and Folke (1999). Hicks *et al.* (2013) focused on coral reefs in Kenya, Tanzania, and Madagascar. The research featured broadly defined cultural/spiritual locations and used semi-structured interviews to collect the data necessary to analyse trade-offs and synergies between this ecosystem service and 7 other ecosystem services without monetary non-market valuation. The perception of ranking, synergies, and trade-offs differed markedly between the different groups included in the study (managers, fishermen, and scientists). The fishermen ranked the cultural service more highly than did the managers and scientists, but had it linked to fewer ecosystem services within the system. Moberg and Folke (1999) present a brief survey of some of the literature related to coral reefs and ecosystem services, briefly mentioning that cultural services include recreation, aesthetics, livelihoods, and “cultural and spiritual values.” These values are not particularly elaborated on or valued within this paper. Therefore, although these two studies do provide some starting points for understanding cultural heritage (and other cultural ecosystem services) in the context of coral reefs, the research is not yet advanced sufficiently to facilitate the treatment of those services within this study.

C3.10 Raw materials

C3.10.1 Seagrass

It is worth noting that fairly recent literature suggests that seagrass beds that produce fibrous debris (such as *Posidonia oceanica*) may be the source of more intentionally utilized raw materials (e.g. for biofuels, for agriculture, as bulking agent, or as a growing media) in the future (Cocozza *et al.* 2011). This implies an ability to increase the provision of ecosystem services by seagrass beds, and to safeguard the provision of other services (such as beach-based recreation), by intentionally looking for uses of seagrass debris that washes ashore. Although this is a valid point, it will not be possible to make assumptions regarding changes in the provision of this service (or cascading effects on other services) in the context of a global MPA expansion scenarios.

C3.10.2 Macroalgae

None found.

C3.10.3 Mangroves

The literature searches conducted returned studies suggesting that harvested mangrove biomass can be used as biosorbents to help remediate terrestrial environments contaminated with heavy metals (Elangovan *et al.* 2008; Oo *et al.* 2009), and mangroves can be the source of broodstock for shrimp farms, with 1 ha for mangroves providing, on average, 08-1.5 *Penaeus monodon* spawners (Ronnback *et al.* 2003).

C3.10.4 Coral reefs

None found.

C4 Discussion & conclusion

The survey of the literature conducted for this report yielded a few quantitative relationships that could, contingent upon the necessary input data being available, be used to estimate at least ballpark changes in the provision of marine ecosystem services in either direct or indirect response to MPA designation. Most of the global or generalized relationships found were theoretical or mathematical in nature, rather than being derived from empirical studies or meta-analyses of empirical studies. This was found to be the case despite there being, in some instances, a variety of site-specific case studies at least proximally relevant to the ES in question. This highlights that there is a need to more systematically consider the existing literature and to compile the diverse sources of data necessary to more deeply evaluate the potential for estimating empirically-based, generalized quantitative relationships of marine ecosystem service provision. As a part of this effort, empirical studies need to continue to improve upon study design and the extent to which confounding variables are both monitored and controlled for. Future efforts that are more focused in nature should also refine the search strategy used in this report, seeking gains in efficiency and also exhaustiveness.

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Appendix C1

*Table C1.1 This table shows the sequence and search strings used to identify the literature needed for this study. The * symbol is the 'wild card' symbol in WOK*

Intended MES link	Search string used	# Added to Marked List
Seafood	Catch* AND Marine Protected Area	109
Ornamental Resources	Seashell AND Marine Protected Area	0
Ornamental Resources ¹	Sea shell AND Marine Protected Area	5
Coastal Erosion Prevention	Kelp AND Erosion	11
Climate Regulation	Kelp AND (Carbon sequestration OR carbon export)	6
Climate Regulation	Macroalgae AND (Carbon sequestration OR carbon export)	11
Waste Treatment	Kelp AND Bioremediation	1
Waste Treatment	Macroalgae AND Bioremediation	26
Waste Treatment	Macroalgae AND Waste	0 ²
Waste Treatment	Kelp AND Waste	8
Lifecycle Maintenance	Kelp AND Nursery	5
Lifecycle Maintenance	Kelp AND Essential Habitat	13
Lifecycle Maintenance	Macroalgae AND Essential Habitat	12
Lifecycle Maintenance	Macroalgae AND Nursery	36
Seafood	CPUE and Marine Protected Area	7
Tourism & Recreation	Kelp AND recreation*	17
Tourism & Recreation	Kelp AND recreational CPUE	0
Tourism & Recreation	Kelp AND recreation* AND CPUE	1
Tourism & Recreation	Macroalgae AND recreation* AND CPUE	0
Tourism & Recreation	Macroalgae AND recreation*	12
Tourism & Recreation	Recreational Fishing AND CPUE AND Marine Protected Areas	0
Tourism & Recreation	Kelp AND Touris*	0
Tourism & Recreation	Macroalgae AND Touris*	18
Tourism & Recreation	Marine Protected Area AND touris*	84
Tourism & Recreation	Coral* AND (recreation* OR tour*)	153 ³
Coastal Erosion Prevention	Coral* AND Coastal Erosion ⁴	43
Coastal Erosion Prevention	Coral* AND Coastal Protection	10

*Table C1.1 This table shows the sequence and search strings used to identify the literature needed for this study. The * symbol is the 'wild card' symbol in WOK (continued)*

Intended MES link	Search string used	# Added to Marked List
Coastal Erosion Prevention	Coral* AND Wave Propagation	9
Coastal Erosion Prevention	Mangrove* AND Wave Propagation	5
Coastal Erosion Prevention	Mangrove* AND Coastal Protection	28
Coastal Erosion Prevention	Mangrove* AND Coastal Erosion	27
Coastal Erosion Prevention	Mangrove* AND Wave Attenuation	4
Coastal Erosion Prevention	Coral* AND Wave Attenuation	6
Coastal Erosion Prevention	Macroalgae AND Wave Propagation	0
Coastal Erosion Prevention	Macroalgae AND Wave Attenuation	0
Coastal Erosion Prevention	Macroalgae AND Coastal Protection	0
Coastal Erosion Prevention	Kelp AND Wave Propagation	1
Coastal Erosion Prevention	Kelp AND Wave Attenuation	8
Coastal Erosion Prevention	Kelp AND Coastal Protection	0
Coastal Erosion Prevention	Seagrass* AND Wave Propagation	6
Coastal Erosion Prevention	Seagrass* AND Wave Attenuation	13
Coastal Erosion Prevention	Seagrass* AND Coastal Protection	2
Coastal Erosion Prevention	Seagrass* AND Coastal Erosion	1
Air Purification	Mangrove* AND Air Pollution	3
Air Purification	Mangrove* AND Particulate Matter	4
Air Purification	Mangrove AND Air Purification	0
Air Purification	Mangrove* AND Atmospheric Pollution	0
Air Purification	Seagrass* AND Atmospheric Pollution	0
Air Purification	Seagrass* AND Air Pollution	0
Air Purification	Seagrass* AND Particulate Matter	2
Waste Treatment	Mangrove* AND Bioremediation	17
Waste Treatment	Mangrove* AND Waste Treatment	3
Waste Treatment	Mangrove* AND Pollution Control	3
Waste Treatment	Mangrove* AND Marine Pollution	6
Waste Treatment	Mangrove* AND Waste	97
Waste Treatment	Seagrass* AND Bioremediation	3

*Table C1.1 This table shows the sequence and search strings used to identify the literature needed for this study. The * symbol is the 'wild card' symbol in WOK (continued)*

Intended MES link	Search string used	# Added to Marked List
Waste Treatment	Seagrass* AND Waste Treatment	0
Waste Treatment	Seagrass* AND Pollution Control	0
Waste Treatment	Seagrass* AND Marine Pollution	3
Waste Treatment	Seagrass* AND Waste	23
Lifecycle Maintenance	Mangrove* AND Nursery	126
Lifecycle Maintenance	Mangrove* AND Essential Habitat	1
Lifecycle Maintenance	Seagrass* AND Nursery	85
Lifecycle Maintenance	Coral* AND Nursery	38
Lifecycle Maintenance	Coral* AND Essential Habitat	2
Climate Regulation	Mangrove* AND (carbon sequestration OR carbon export)	71
Climate Regulation	Seagrass* AND (carbon sequestration OR carbon export)	19
Tourism & Recreation	Mangrove* AND (tour* OR recreation*)	20
Tourism & Recreation	Seagrass* AND (tour* OR recreation*)	20
Raw Materials	Seagrass* AND Raw Material	1
Seafood	(Marine Protected Area OR Marine Reserve) AND CPUE	17
Lifecycle Maintenance	(Marine Protected Area OR Marine Reserve) AND Nursery	27
Seafood	(Marine Protected Area OR Marine Reserve) AND (Spill over and spillover)	8
Tourism & Recreation	(Marine Protected Area OR Marine Reserve) AND (tour* OR recreation*)	79
Climate Regulation	(Marine Protected Area OR Marine Reserve) AND (carbon sequestration OR carbon export)	0
Waste Treatment	(Marine Protected Area OR Marine Reserve) AND Waste	0
Coastal Erosion Prevention	(Marine Protected Area OR Marine Reserve) AND (Erosion OR wave propagation OR wave attenuation OR coastal protection)	3
Cultural Heritage and Identity	Seagrass* AND Cultur*	12
Cultural Heritage and Identity	Mangrove* AND Cultur*	15
Cultural Heritage and Identity	Coral AND Cultur*	30
Aesthetic Information	Seagrass* AND Aesthetic	0
Aesthetic Information	Mangrove* AND Aesthetic	1
Aesthetic Information	Coral* AND Aesthetic	11
Aesthetic Information & Cultural Heritage and Identity	(Kelp OR Macroalgae) AND (Aesthetic OR Cultur*)	0

*Table C1.1 This table shows the sequence and search strings used to identify the literature needed for this study. The * symbol is the 'wild card' symbol in WOK (continued)*

Intended MES link	Search string used	# Added to Marked List
Tourism and Recreation	Sea turtle AND (Tour* OR Recreation*)	26
Cultural Heritage & Identity	Sea turtle AND Cultr*	12
Biological Control	Sea turtle AND Biological Control	0
Gene Pool Protection	Sea turtle AND gene pool	0

¹ This search yielded results more relevant to the seafood service than to ornamental resources

² None of these studies were considered further because they focused on biofuels in a way that was not relevant to this study

³ The proportion considered further for this search is quite small because the same kinds of themes were reiterated frequently, and as this purview of this review was not to be exhaustive, it was not necessary to include every case study related to, for example, the negative impacts of divers on coral reefs.

⁴ This search was conducted with respect to coastal erosion to exclude the large number of articles discussing the erosion of corals