

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
		No		3.2.1-3.2.7
Recreation & Tourism	Seagrass beds	No		3.7.1
	Macroalgae Stands	No		3.7.2
	Mangrove Forests	No		3.7.3
	Coral Reefs	No		3.7.4
Climate Regulation	Seagrass beds	Yes	131	3.3.1
	Macroalgae Stands	Yes	132	3.3.2
	Mangrove Forests*	Yes	134	3.3.3
	Coral Reefs	No		3.3.4
Erosion Prevention	Seagrass beds	Unclear ¹		3.4.1
	Macroalgae Stands	No		3.4.2
	Mangrove Forests	No		3.4.3
	Coral Reefs	No		3.4.4
Waste Treatment	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 mangrove 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.

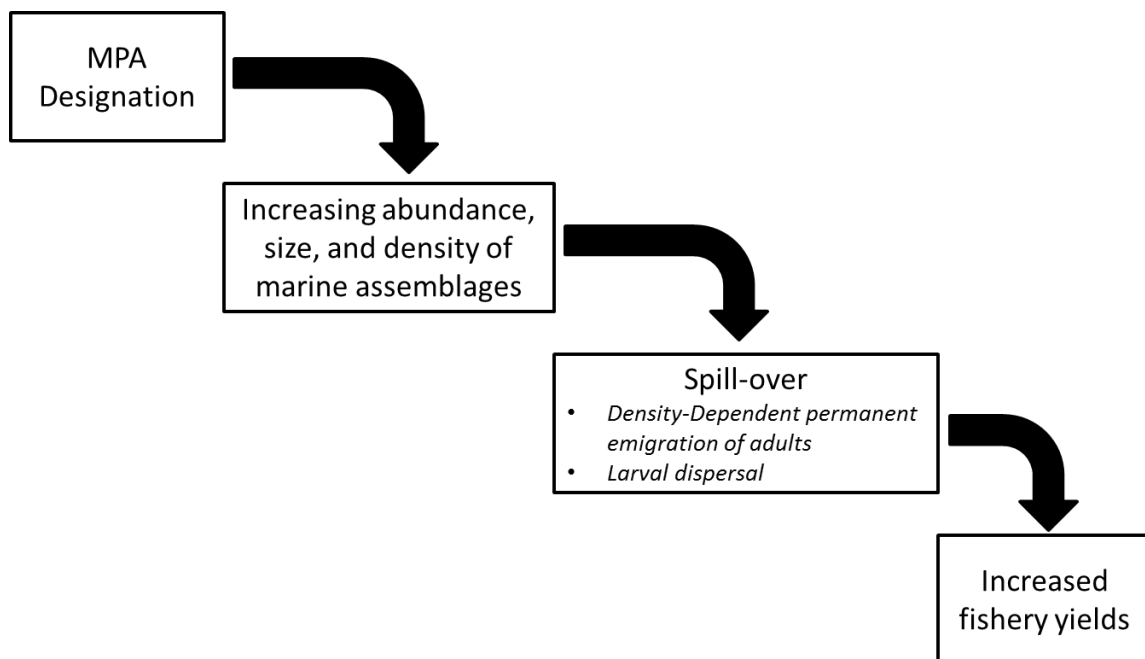


Figure C1 Anticipated cascade from MPA designation to improvements in fishery yields

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 a 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).

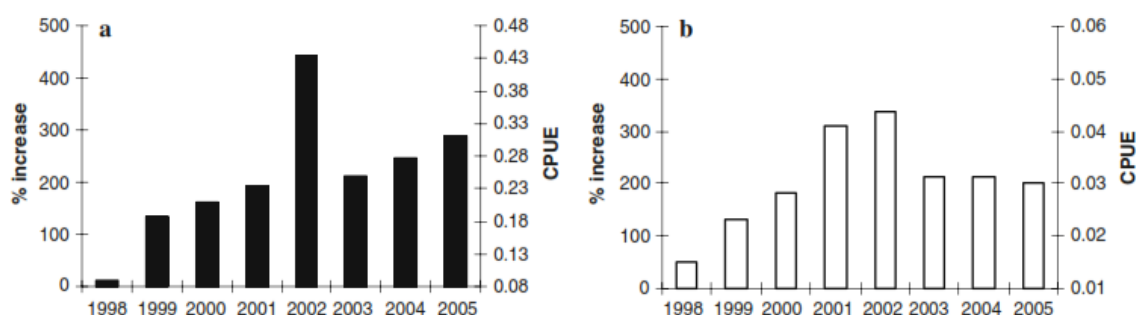


Figure C2 This figure is originally figure 2 from Follesa *et al.* (2008), where it has the caption "Rate of increase (%) of *Palinurus elephases* abundance inside (a) the reserve area and in the surrounding zone (b)

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.

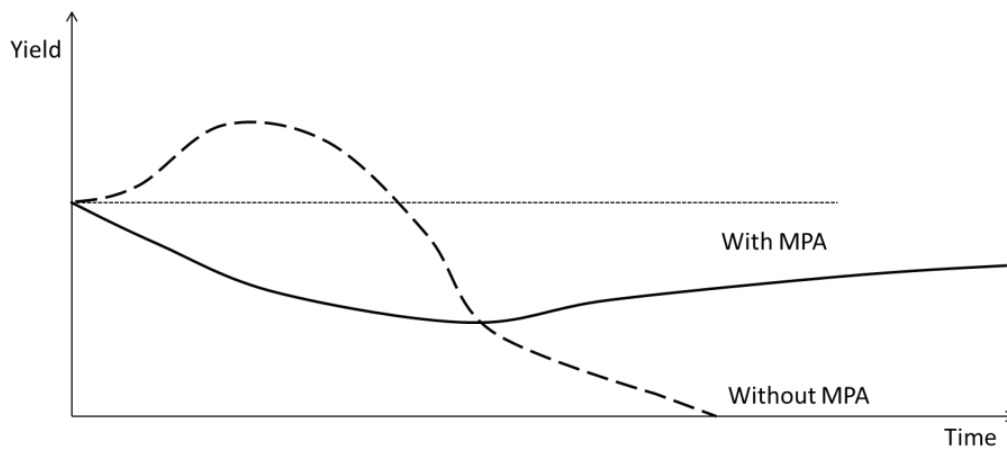


Figure C3 Hypothetical yield from an overharvested fishery with time both with and without an MPA where the absence of a counter-factual scenario changes the interpretation of MPA impact data

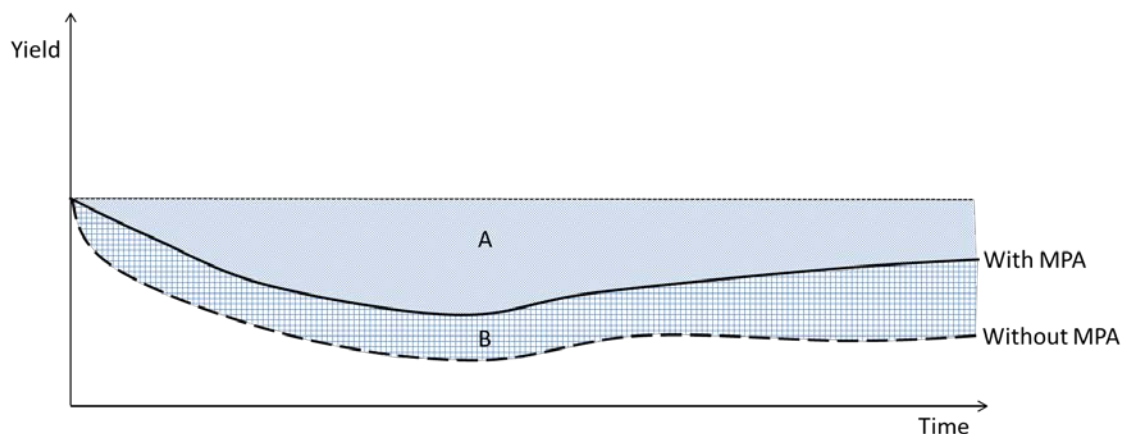


Figure C4 Alternative hypothetical yield from an overharvested fishery with time both with and without an MPA where the absence of a counter-factual scenario changes the interpretation of MPA impact data. Zone A represents the loss that would be perceived if the impacts of the MPA were measured relative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

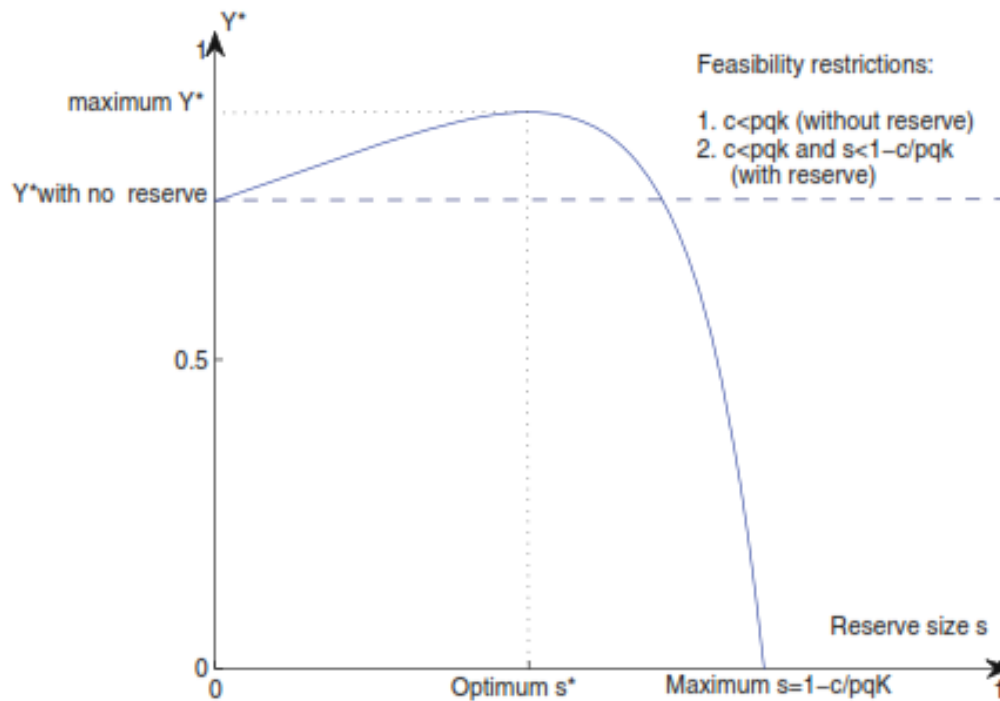


Figure C5 “The relationship between harvest and reserve size using [the] reduced system. Catches increased with MPA size reaching a maximum. The parameter values are: $r=0.9$, $c=0.6$, $p=1$, $q=0.5$, $K=4\dots$ ” Caption and figure from figure 2 from (Bensenane et al. 2013)

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):

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

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 undertaken 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 of 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 Cyr ⁻¹)	Buried (globally) inclusive of sediment trapped by seagrass beds		
Global	Mixed (un-specified)	4.2-8.4 Pg C	Global pool of organic carbon contained in/by seagrass beds	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)
		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		
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 plant 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 yr ⁻¹	Potential emissions from mangrove forest deforestation	The potential emissions can exceed the storage capacity of these forests	
Global data	Does not specify (Unspecified all species)	163 g OC m ⁻² yr ⁻¹ 95% CI: (132, 204) 26.1 Tg OC yr ⁻¹ 95% CI: (21, 32.4)	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	Breithaupt <i>et al.</i> (2012)
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		
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	Wang <i>et al.</i> (2013)
	<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 gymnorhiza</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* bannette 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 mariqueter* (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 difference 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)] = 100µg/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 (Lamb *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' in 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 metals 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 (*Tautogolabrus 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*, *Spondiyosomo cantharus*, *Diplodus annualaris*, and *Dicentrachus 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	<p>Catch-Seagrass-Effort models⁴³</p> <p>All the models estimated were significant at the 0.05 level and featured R² values between 60% and 97%.</p> <p>The catch estimates were further decomposed into commercial, recreational, and discard by using the following estimated relationships:</p> $C_{total,i} = C_{com,i} + C_{rec,i} + C_{dis,i}$ $C_{rec,i} = 0.25C_{com,i}$ $C_{dis,i} = 0.286C_{com,i}$ <p>Assumptions:</p> <p>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</p>	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 (Clynick 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 λ_i 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 or 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 sandstone 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 that 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