University of Wollongong

Research Online

Faculty of Law, Humanities and the Arts - Papers

Faculty of Arts, Social Sciences & Humanities

January 2018

MPAs, fishery closures and stock rebuilding

Richard Kenchington
University of Wollongong, rkenchin@uow.edu.au

Michel J. Kaiser
Bangor University

K Boerder
Dalhousie University

MPAs, fishery closures and stock rebuilding

Abstract

Few regions on earth remain untouched by fishing activity. As such, effective longterm no-take marine reserves (NTMRs) and other area-based management systems that restrict fishing serve as vital reference areas for assessing the magnitude and recovery potential of marine ecosystems from human influence over both space and time. Much of the peer-reviewed literature and meta-analyses demonstrate variable but overall positive responses to such protection for fish and some other taxa. These include significant increase of stock species abundance and biomass within boundaries, cross-boundary spill-over of adults and larvae, and increased egg production. A reserve may meet its biodiversity objectives and contribute to fished stocks, however, determining the extent of that contribution depends on several factors. These factors include: understanding and monitoring the species biology, and the effectiveness of and compliance with management, and changes in fishing pressure external to the NTMR or other type of spatially-managed area from which some fishing activities are excluded. This presents significant data and cost challenges of scope, scale and monitoring method for assessment of NTMRs and fishery management throughout the, often extensive, range of the stock. The situation is one in which proof is hard to establish and absence of proof of effect is not proof of absence of effect (the issue of Type I and Type II errors in experimental design).

Publication Details

Kenchington, R., Kaiser, M. J. & Boerder, K. (2018). MPAs, fishery closures and stock rebuilding. In S. M. Garcia & Y. Ye (Eds.), Rebuilding of Marine Fisheries Part 2: Case Studies (pp. 182-205). Rome: Food and Agriculture Organization of the United Nations. http://www.fao.org/3/ca0342en/CA0342EN.pdf

MPAs, fishery closures and stock rebuilding

Kenchington[1] R., Kaiser[2] M.J. and Boerder[3] K.

[1]*Australian National Centre for Ocean resources and Security, University of Wollongong, NSW 2522 AUSTRALIA*
[2]*School of Ocean Sciences, Bangor University, Menai Bridge, Anglesey, United Kingdom*
[3]*Department of Biology, Dalhousie University, Nova Scotia, Canada*

Abstract

Few regions on earth remain untouched by fishing activity. As such, effective long-term no-take marine reserves (NTMRs) and other area-based management systems that restrict fishing serve as vital reference areas for assessing the magnitude and recovery potential of marine ecosystems from human influence over both space and time. Much of the peer-reviewed literature and meta-analyses demonstrate variable but overall positive responses to such protection for fish and some other taxa. These include significant increase of stock species abundance and biomass within boundaries, cross-boundary spill-over of adults and larvae, and increased egg production. A reserve may meet its biodiversity objectives and contribute to fished stocks, however, determining the extent of that contribution depends on several factors. These factors include: understanding and monitoring the species biology, and the effectiveness of and compliance with management, and changes in fishing pressure external to the NTMR or other type of spatially-managed area from which some fishing activities are excluded. This presents significant data and cost challenges of scope, scale and monitoring method for assessment of NTMRs and fishery management throughout the, often extensive, range of the stock. The situation is one in which proof is hard to establish and absence of proof of effect is not proof of absence of effect (the issue of Type I and Type II errors in experimental design).

The response of marine biota to the implementation of a fishery closure typically take 3 to 5 years to appear. In contrast, the loss of access to fishing opportunities is immediate and is particularly challenging for fishers with high dependence on it. This raises the need to engage with stakeholders to address issues such as displacement of effort to other areas already subject to fishing and may require compensation or support for development of alternative economic activities to achieve effective temporary or permanent removal of fishing effort. In situations where the benefits from closures are uncertain, scientists, planners and managers need to express due caution in forecasting their contribution to stock rebuilding. Unrealistic expectations and unrealised benefits harm relationships with stakeholders and undermine future management relationships that depend on trust.

Performance evaluation of NTMRs or other area-based management measures in relation to the different primary objectives of conservation of fish stocks and holistic conservation of marine biodiversity presents data challenges. These challenges are exacerbated in the broader multi-sectoral context of blue economy uses and land-sourced and atmospheric impacts that affect marine biodiversity. The scope, scale and precision of data required for the management and evaluation of fish and shellfish

stocks throughout their range differ from the data-reporting required to evaluate and achieve holistic biodiversity conservation within protected areas and networks of protected areas. This mismatch limits the capacity to discriminate the effect of a protected area on stocks that function at large spatial scales. A convergence of the conservation and fisheries science communities is required to scale data collection appropriately to understand more fully the contribution of protected areas to the conservation of commercially exploited species.

1. BACKGROUND

For most of history prior to the 19th century, fisheries have been locally based, seasonal and often varied considerably according to longer-term natural variations in species availability. Consequently, there was a prevailing understanding that the scale of human activities was unlikely to have significant impacts at an oceanic scale. Historical access (e.g. exploitation of cod on the Grand Banks in the 15th century) of distant fish stocks was constrained by vessel capability, the high risks of seafaring and challenges of product preservation (Jennings et al., 2001). Late 19th century concern at the decline of fish stocks in the North Sea led in the 20th century to the development of marine science, and concepts of management for conservation of fish stocks and more recently biological diversity.

The vulnerability of marine ecosystems to the application unsustainable human pressures is increasingly evident. Wild caught fisheries peaked in terms of landings by the 1990s followed by a decline to c 80 Mt at present (FAO, 2016). Natural refuges have reduced because of the increases in the range and scale of fishing capacity, precision of navigation and fish finding technologies, and degradation of habitat as a secondary effect of fishing. There is also an increasing range of economic uses of marine space and coastal lands that impact (but do not depend on) on the productivity or well-being of marine ecosystems (e.g. land-based pollution, coastal developments, shipping, renewable energy installations, mining). The links between biodiversity and ecosystem services such as primary and secondary production (including fisheries production) are supported by a variety of lines of evidence (see Worm et al., 2006). For these reasons, there has been a steady convergence of fisheries and biodiversity conservation management objectives over the last 20 years. Hence it is no surprise that some of the tools used by managers to achieve these management objectives have considerable overlap, e.g. the use of area closures or fishing gear restrictions.

The concept of using closed areas to limit human activities is not new. Customary management of local fisheries includes examples of the use of permanent, seasonal and temporary closures (e.g. Johannes, 1981). Nevertheless, the more recent concept of closures for the protection of biodiversity has often generated conflict between fishing and conservation interests. This is arguably a distraction from the shared interest in sustaining biodiversity and food production in the context of increasing human activities and impacts affecting marine space.

In this chapter we present a summary of the state of knowledge and challenges of assessing the effects that areas closed to fishing may have in terms of fish stock rebuilding irrespective of whether the primary objective of the closures was biodiversity conservation. We provide brief overviews (but not a comprehensive survey) of reported outcomes of the utilisation of different categories of marine protected areas that exclude fishing through no-take marine reserves (NTMRs) or other management systems that include area closures or preclude some form of fishing from a specified area of the sea. NTMRs are also referred to as No-Take Zones, or MPAs categories I and II, as defined by the International Union for the Conservation of Nature (IUCN) classification (Day et al., 2012) and they exclude all extractive forms of activity (e.g. fishing).

2. POLICY DRIVERS AND INEVITABLE TENSIONS

The primary objectives of fish stock and biological diversity conservation differ but both depend on the wellbeing of the marine environment in the face of the increasing range and extent of human uses and their associated impacts on marine ecosystems. The extent to which a spatial closure may contribute to fish or shellfish stock maintenance or rebuilding depends on the biology of the target species; the management of the fishery throughout its range and the ability to achieve user or stakeholder engagement and compliance with fishery and MPA measures.

The international framework for conservation of marine fish stocks and biodiversity is provided through the United Nations Convention on Law of the Sea (LOSC) and the United Nations Convention on Biological Diversity (CBD).

- Article 61 of LOSC refers to conservation of “living resources through measures designed to maintain populations of harvested species at levels that can maintain the maximum sustainable yield”. The primary objective of fisheries management is to achieve maximum sustainable yield of living natural resource stocks;
- Article 8 of CBD identifies protected areas as a primary means of addressing conservation of biological diversity. The primary objective is conservation of biodiversity which is defined as: “diversity within species, between species and of ecosystem” at levels from populations, species and communities to ecosystems.

The IUCN definition of a marine protected area is: *A clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values.* The IUCN has six broad categories of protected area (Dudley, 2008).

Day et al. (2012) have provided guidance on how IUCN categories should be applied and interpreted in the marine context of sustainable use, cross-boundary and multi-directional linkages of the seabed and water column. For all practical purposes, Marine Protected Areas (MPAs) are spatially defined areas in which human activities are more explicitly controlled than outside them. Impact assessment and specific conditions are typically required for a new or altered form or level of activity. For any activity the control may be defined within a spectrum ranging from <u>total prohibition</u> of any use, to specific uses <u>allowed with permit</u>, subject to environmental impact assessment and consequent conditions, and to uses that are <u>allowed subject to any general constraints</u> or operational conditions defined in regulation.

The most restrictive conditions exclude fishing in IUCN MPA categories I and II or NTMRs (also referred to as No-Take Zones). Category I excludes all access and activities other than for approved research and management actions while category II provides for access and enjoyment of areas free from extractive activities. Fishing activities with significant habitat impact such as trawling and dredging are excluded from reserves aimed at habitat protection (IUCN category IV). Multiple- use areas (Category VI) are consistent with the overall conservation objective that provides for verifiably sustainable human use and impacts. Such management systems often include buffers zones with higher levels of protection than other parts of the same system. The term MPA is often used as a generic name for NTMRs, creating regrettable confusion.

The different concepts of conservation implicit in international targets for restoration of fish stocks and marine biodiversity conservation create further regrettable confusion. Targets have been set and extended since the 2002 WSSD Plan of Implementation (§31 and 32). The most recent reiteration was contained in United Nations Sustainable Development Goal 14 (United Nations, 2016):

- *14.2 By 2020, sustainably manage and protect marine and coastal ecosystems to avoid significant adverse impacts, including by strengthening their resilience, and take action for their restoration in order to achieve healthy and productive oceans;*
- *14.4 By 2020, effectively regulate harvesting, and end overfishing, illegal, unreported and unregulated (IUU) fishing and destructive fishing practices and implement science-based management plans, to restore fish stocks in the shortest time feasible at least to levels that can produce maximum sustainable yield as determined by their biological characteristics;*
- *14.5 By 2020, conserve at least 10 percent of coastal and marine areas, consistent with national and international law and based on best available scientific information.*

The target of conserving 10% of coastal and marine areas reiterates the CBD Aichi Target 11 of the UN Convention on Biological Diversity (CBD, 2011)[1]. This target is not strictly about MPAs, as it calls for the consideration of "*other effective area-based conservation measures*" This definition opens the possibility to assess the contribution to the conservation of biodiversity that could be made by implementing spatial area closures as fishery management tools. The use of closures in fish stock conservation mostly addresses protection of fish stocks at vulnerable life cycle stages and is designed to minimize and mitigate the collateral impact of fisheries on biodiversity.

The use of NTMRs in biodiversity conservation provides for holistic protection of marine biodiversity in precautionary networks of healthy representative marine ecosystems to provide resilience in the face of increasing human impacts and environmental change. Typically, there is competitive tension between the social, economic and political drivers of fish stock and biodiversity management systems. Depending on context, a closure intended to achieve one objective may have positive, neutral or negative effects for the other. The different shades of meaning of "conservation" reflect constructive ambiguity in developing text to address different and potentially conflicting sectoral priorities.

For the purposes of this chapter, we focus primarily on the performance of NTMR areas that are closed for long periods of time (years) and their apparent roles in fish stock rebuilding irrespective of whether or not the primary objective of the closures was biodiversity conservation or fishery management. Whilst our main focus has been on the usefulness of NTMRs, we have also examined the use of habitat protection zones within MPAs that eliminate bottom gears but enable continuation of other fisheries. We explored this literature to understand to what extent these areas can have significant benefits for conservation of benthic biodiversity (Sciberras et al., 2015)

3. DATA CHALLENGES FOR PERFORMANCE ASSESSMENT OF NTMRS IN FISH STOCK AND BIODIVERSITY CONSERVATION

The performance criteria, management arrangements and monitoring of MPAs are elaborated to address the primary objective and scale of holistic biodiversity protection of a site or network. They differ markedly from those of fisheries surveys that are designed to assess population status across much larger geographic areas and across

[1] Target 11: By 2020… 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas <u>and other effective area-based conservation measures…</u>

time. Thus, the different sampling approaches create inevitable difficulties when trying to determine changes in population status that might be attributable to the use of MPAs or other spatial restrictions.

Effective assessment of the performance of MPAs worldwide is constrained by the high proportion of poorly resourced and inadequately enforced "paper parks". Edgar et al. (2014) used an extensive database to investigate the ecological response of fish communities within MPAs and concluded that when there was an absence of evidence of recovery of fish biomass, in the case of many MPAs this could be attributed to ineffective MPA compliance and high spatial variability in fish densities. It is also noteworthy that much of the literature has tended to focus on the responses of "fish" because these are most often the focus of exploitation and hence the group expected to respond most strongly to protection from exploitation. This focus on fish means that we have far fewer insights into the responses of a wider range of taxa at lower trophic levels that might benefit indirectly from protection (Stewart et al., 2009).

Despite these limitations, much of the peer-reviewed primary literature and meta-analyses demonstrate overall positive responses of the studied taxa to reserve protection on the biomass, numerical density, species richness, and size of fished target species (most often fish species) within the boundaries of NTMRs. These responses were recorded over periods from several years to decades (e.g. Abesamis et al., 2006; Lester et al., 2009; Stewart et al., 2009; Russ et al., 2015; Sciberras et al., 2015). There is substantial evidence that, with high compliance, NTMR closures of both reef and coastal fish and shellfish fisheries can lead to a demonstrably significant increase in population and biomass of fishery target species within the NTMR, in some cases after a period of several or more years (e.g. Stewart et al., 2009). This increase in abundance and biomass continues thereafter until carrying capacity is reached which depends on the species and the environmental context (Kaiser et al., 2018). There is evidence of increased larval and juvenile settlement and adult recruitment outside reserves (otherwise known as spillover) of fish providing an increase in CPUE or body-size near NTMRs (McClanahan & Mangi, 2000; Cudney-Bueno et al., 2009) and of increased egg production per unit area of reserve (Beukers-Stewart et al., 2006; Kaiser et al., 2007).

The assessment of the effect of NTMRs and fishery closures on a fish stock requires consideration of the biology and history of management of the target fish species in the protected area and throughout its range. There are two ways in which NTMRs may contribute to fished stocks: (i) post-settlement and adult spillover; and (ii) net export of eggs and larvae from accumulated biomass of reproductively mature adults within the reserve. In contrast to this rather 'area-specific' approach, fish stock management demands a more complex 'whole stock' approach with comprehensive sampling across wide geographic areas of population status (age, size-distribution, abundance) coupled with estimates or quantification of natural and fishing mortality. This mismatch in the scaling of the science to address these two different questions means that it is difficult to understand how one management objective impacts upon the other (either positively or negatively).

The monitoring conducted to inform effective fisheries management focuses primarily on total harvest, catch per unit effort (CPUE) and, as required, on by-catch of non-target species. Long standing fisheries have typically existed and grown with new demands and technical capacity and have thus been managed with no pre-fishery baseline or control data. New fisheries may start with exploratory fishing and expand ahead of substantial scientific evaluation of stock or environmental sustainability.

Science-based assessment for fishery management is thus constrained by the absence of unfished control sites (Papworth et al., 2009) and the phenomenon of shifting baseline due to discontinuous records (Pauly, 1995).

Determining the effect of a NTMR or a network of NTMRs on a fishery resource should be informed by (but in many cases is typically constrained by limitations of) detailed area-based data on the management of the fisheries and its performance on the target stocks and other species richness, abundance and distribution throughout the resources range.

A further data constraint may emerge on legal grounds of commercial confidentiality of high-density fishing data that could otherwise inform on the relative significance of areas of the stock range in terms of CPUE and cost per unit effort. Aggregating data at coarse spatial scales to preserve confidentiality makes it difficult to identify the broader effects of a positive fish stock signal from NTMR monitoring in the noise of fishing data throughout the population range (Hinz et al., 2013).

The limitation of fisheries data may be partly compensated by modelling based on approximations and assumptions from backward-projection of stock size under zero-fishing conditions and from similar fisheries in similar situations. These approximate comparators are imperfect for the target stocks or well documented non-target stocks and are constrained by the underlying assumption of ecosystem stability with respect to past and projected climatic conditions, population traits, and on the level of compliance with management measures. When considering the extent to which we could determine the effects of a reserve on wider population status, it is important to acknowledge that that model predictions often have such large measures of uncertainty that the effects of any single reserve or network on population status would likely be 'lost' within the boundaries of error within the estimated projections. It is sufficient to acknowledge that it remains difficult to achieve precise assessment of the large-scale comprehensive impacts of fisheries and their interactions with other human uses and impacts) affecting marine ecosystems, including of climatic oscillations and change.

The criteria for monitoring the performance of measures designed to achieve holistic conservation of marine biodiversity usually focus on community metrics such as species abundance, extent, biomass and diversity. Adequate assessment of the performance of these metrics is complicated by the lack of reference areas against which to assess them. Often the primary focus on establishing representative areas of habitats and communities that are protected from impact is to enable them to act as control or reference sites against which the effects of direct human uses may be discerned against the combined effects of system dynamics and indirect human impacts from coastal run-off and atmospheric change.

Russ (2002) identified five measurable indicators of the status of fish community status within the boundaries of a reserve:

1. Increased density of target species;
2. Increased biomass of target species;
3. Larger mean size/ increased age of target species;
4. Greater production of propagules (eggs/larvae) of target species per unit area; and
5. Lower or no fishing mortality (F) within reserve sites.

The first three of these indicators can be assessed through biological monitoring. The fourth indicator is usually inferred from the relationship between abundance, body size and propagule production which is sometimes coupled with oceanographic modelling of propagule dispersal (e.g. Kaiser et al., 2007; Carter et al., 2017). The issue of quantifying fishing mortality (F) is more complex due to the challenges of acceptance of closure and robust monitoring to assess levels of compliance. Poaching, illegal fishing and inadequate reporting of catch are frequently mentioned as constraints to assessing the effects of closure. In addition, while F may be reduced or eliminated within the boundaries of a reserve, the net effect on F at a population level may be neutral or even negative if fishing pressure outside the reserve increases (including through effort displacement) or if other human impacts or environmental change affect recruitment. Often, existing fisheries surveys (where they exist) will have their own objectives (e.g. measuring recruitment or overall biomass distribution and structure) and hence be spatially mismatched –in terms of data location and scale of sampling - with the objective of measuring increases of F specifically within areas external to a particular reserve.

Limitations of data on species richness, abundance and distribution may be partly compensated by modelling that is based on inferences derived from seabed mapping and limited sampling of fish and larger invertebrates and surrogacy based on assumptions and extrapolations from knowledge of habitat dependencies in similar situations. Multibeam sonar seabed mapping can be used to describe the physical properties of the seabed which is then related to ground-truthing samples. The downside of such an approach is that the collection and processing of samples of the seabed and associated species is time consuming and resources are typically limited for subsequent analysis, particularly of small sized and soft bodied species that can be important food for fished species (Kenchington and Hutchings, 2012).

In well-resourced NTMR, the effects of management and science programs may be monitored or inferred within the boundary. However, a robust understanding of the broader performance of an NTMR in relation to fish stock rebuilding or overall maintenance of biodiversity depends on many other factors. These include the biology and life-history of the target species, inter-annual environmental variation, adequacy of and compliance with fishery management, and other human impacts throughout the species adult and recruitment range.

4. PERFORMANCE

Adequate management and monitoring programmes may be able to demonstrate performance of increased biomass of species within a well-managed NTMR. Outside the NTMR, this may be accompanied by a reduction, increase or no change in biomass of target and non-target species outside the protected area (e.g. Dinmore et al., 2003). Attributing cause and effect of the contributions of NTMR, fishery management, and factors beyond either management system is challenging. **Table 1** summarises possible causal mechanisms for observed stock biomass performance.

Against this background we present a collection of metanalyses of reserve performance such as: (i) closures in small scale fisheries; (ii) the experience of Great Barrier Reef Marine Park; (iii) closures in demersal trawl fisheries; (iv) fisheries closures for Highly Migratory Species; and (v) examples of the contribution of fishery restricted areas to biodiversity conservation.

TABLE 1
Summary of the response of the status of the population of fished species coincidental with the effective implementation of a no-take marine reserve (NTMR) with an explanation of possible causal mechanisms

Indicator	Possible causal mechanisms
Increased stock biomass	• Spillover of adults and juveniles from NTMR; • Larval export from NTMR; • Better management of fishing effort; or • Strong overall recruitment, unrelated to NTMR.
Stable stock biomass	• Spillover of adults and juveniles from NTMR • Larval export from NTMR • No detectable impact of NTMR, or • External factors offsetting NTMR spillover, or larval export such as impacts of land sourced pollution, other uses of marine space or climate change, compensatory increase in fishing effort
Reduced stock biomass	• No or inadequate spillover of adults and juveniles from NTMR • No or inadequate or larval export from NTMR • Poor overall recruitment unrelated to NTMR • Unsustainable fishing effort outside the NTMR, or • External factors such as impacts of land sourced pollution, other uses of marine space, climate change or predator prey trophic interactions leading to natural decline in background stock biomass

4.1 Meta-analyses of reserve performance

Sciberras et al., (2015) and Halpern (2003) noted that the empirical literature on marine reserves uses a wide range of methodologies to quantify the effects of reserves on specific taxa, and moreover that the characteristics of reserves and control sites can differ dramatically, and both can be affected by inter-annual and decadal oscillations or stochastic impacts (e.g. Stephenson et al., 1970). Both issues complicate comparisons of the performance of different MPAs. In particular, the most common shortcoming of comparative studies of the outcomes of MPAs is the habitat confounding effect and lack of adequate comparator sites for MPAs (Stewart et al., 2009). Furthermore, while there are numerous meta-analyses that attempt to synthesise general patterns from this diverse array of studies (Claudet et al., 2008; Lester et al., 2009; Sale et al., 2014; Sciberras et al., 2015), often the conclusions of such meta-analyses are weakened due to lack of rigour in the methodological approaches adopted within the review process which leaves them prone to bias (see Woodcock et al. 2017). Many of the more widely cited meta-analytical papers on the performance of marine reserves achieved only medium scores for rigour which is a concern if policy makers use these analyses to inform policy development. Nevertheless, despite the potential for bias in some meta-analyses published to date, the conclusions are broadly similar across all syntheses.

Sciberras et al. (2015) compared the response of target and non-target species to the use of NTMRs and areas with partial protection from fishing. They concluded that the response to protection was strongest (and positive) for target species and that NTMRs resulted in larger positive responses than partial protection areas. Stewart et al. (2009) examined the response of target species to NTMRs in temperate waters and found significant and positive increases in abundance, biomass and diversity. However, they noted that most studies were confounded by habitat effects (i.e. when the comparator site is not similar to the manipulated site (in this case the NTMR) and that there was potential bias in the reporting of the response in biomass data. Halpern (2003) provided a review of 89 empirical studies of the effect and the influence of size of marine reserves. Whilst the study was challenging because of differences in the environmental and geographic context, design and methodology of studies and in many cases lack of rigorous standards of broader scientific literature, the overall conclusion suggested that reserves were generally effective. Gruss et al. (2014) reviewed results of empirical studies and reports of the conservation and fishery effects of 28 NTMRs specifically designed to protect the spawning aggregation sites of targeted species for periods ranging from 3 to 38 years. Whilst 12 had positive results for biodiversity conservation effects, 10 had negative results, 5 showed no clear change and one had no data. There was no information for fishery effects in 17 of the studies, 3 results were positive, 3 negative and 5 showed no clear change. The "non-positive direct conservation effects" of Fish Spawning Aggregation (FSA) protected areas were attributed mostly to poor or non-existent internal enforcement; small size of the FSA; exploitation of fish in their migration routes to the FSA; and absent or ineffective measures to control fishing mortality during non-aggregating times.

While the general conclusion from most of the published analyses indicates a consistent positive outcome following the implementation of NTMRs and partial protection areas, the response in individual cases is strongly variable and there is a lack of data on the intensity of resource exploitation and compliance as shown in the analytical outputs in Stewart et al. (2009).

4.2 Small-scale coastal fisheries

4.2.1 Changes in community metrics

Recognition of the need for marine conservation in the late 1970s – 1980s coincided with the publication of studies undertaken on the West Pacific, South East Asia, Japan and northern Australia of surviving traditional practices of coastal fishery management through tenure, cultural rights, closures or taboos (e.g. Johannes, 1981; Ruddle, 1988). Many of these traditional practices had been displaced or weakened by colonial rule or were being overwhelmed by economic development or population growth. In the islands and coastal areas of South East Asia and the Western Pacific growing population combined with falling yields and low income from traditional fishing activities around the coral reefs provided perverse and damaging incentives to use more efficient (but destructive) fishing methods including explosives, poisons, muro-ami push nets, small-mesh nets, and spear fishing using SCUBA (Russ et al., 2004).

A research and community engagement program supported and monitored by Siliman University in the Philippines enabled the establishment of a NTMR at Apo Island in 1974 and Sumilon Island in 1982 with long-term monitoring of the outcomes of these management measures inside and outside the reserves. Importantly, the program drew on traditional practices and developed additional economic opportunities to enable participation and empowerment of community-based comparisons of reserve and non-reserve biomass of coral reef fished species and total catch. The Apo Island observations included a serendipitous replication when the reserve area was re-established after a governance change had re-opened the reserve area to fishing for 3 years.

Alcala and Russ (2006) reported a 3.1-fold increase of all target fish over a 9-year period at Sumilon Island and 4.6-fold increase over an 18-year period at Apo Island. For the larger predatory fish, the increases were respectively almost 12-fold and 17.3-fold over 9 and 18 years respectively. There was a 3 to 17-fold increase in biomass of fished species within the reserves. Outside the reserves there was no overall increase of biomass. A similar outcome was reported by Guidetti et al (2008) from visual surveys undertaken in 15 MPAs in Italy. They found that total fish density in reserves was on average 1.15 times great than in fished areas noting that the effects of protection on fish species and trophic groups were not detected in unenforced reserves.

In a well-monitored MPA (the Bamboung bolong, in Senegal) surveyed in 2003, closed by fishermen in 2004, and monitored for 5 years, total biomass, maximal fish length, number of species, percentage of large iconic species increased. The community structure was modified, with more small fish, more big fish (new large species and more large individuals in the original species) and fewer medium sized fish. The trophic structure was modified with an overall increase of the mean trophic level, resulting from an increase of the percentage of generalist or piscivorous predators and a sharp decrease of herbivorous and detritivores and other low trophic level species. The marine predators which numbers and size were reduced by fisheries became again important components of the protected system (Ecoutin et al., 2014).

4.2.2 Changes in measures of catch per unit effort

There are relatively few experimental studies of the effects of MPAs on the CPUE in surrounding fished areas. Alcala and Russ (2006) quantified changes in CPUE for Sumilon Island and Apo Island. They found that higher numbers of fished species were recorded within 200 m of the reserve boundaries together with a 50% increase in CPUE and a 46% reduction of total fishing effort. These observations suggest that a spillover effect had occurred because of the implementation of the marine reserve. The long-term outcome of the implementation of these reserves demonstrated that closure of 10 to 25% of fishing area of Apo and Sumilon islands improved CPUE and did not reduce the total fish yield at either island (Alcala and Russ, 2006). Guidetti and Claudet (2010) reported on a 3-year experimental study co-managed with fishers to compare CPUE in one reserve with an MPA buffer zone that was opened after 5 years of enforced protection and then compared with the CPUE in the surrounding fishing ground. The objective was to maintain catch levels through effort control and by using specified fishing methods and effort intensity. The outcome after three years of co-managed exploitation was a level of CPUE that was approximately double that which occurred outside the MPA.

In South Africa (Kerwath et al., 2013) studied a boat-based commercial fishery to compare CPUE and total catch trends in 5 nautical mile grid squares of three south coast fishing regions with those close to the Goukamma marine reserve. This demonstrated increases in population and biomass of target species in the reserve and a doubling of off-reserve CPUE over five years near the NTMR boundaries suggesting adult spillover and larval dispersal for a resident species with occasional home range relocations unrelated to density dependence. In a similar study, Blyth-Skyrme et al. (2004) used records of trophy fish catches by sport fishers to ascertain changes in the mean size of trophy fish of a range of species that varied in life history. They compared catches in large geographic areas adjacent to and immediately surrounding a 350 km^2 partial protection area. For species with relatively early age at maturity, the mean size of trophy fish of all species recorded in the study remained stable or increased close to the partially protected area but decreased away from it. For species with late age at maturity no impact on trophy size was noted. This may mean that short-lived animals reacted positively while long-lived ones did not, at least in the time-frame of the experiment. However, this remains speculative in the absence of more formal tagging studies to confirm this hypothesis. The latter is a particularly insightful study as it indicates the importance of considering fish life history characteristics when considering the size and configuration of marine reserves and seeking to understand and compare the effects of NTMRs on fish stocks or biodiversity.

4.3 Great Barrier Reef Marine Park – a case study

The Great Barrier Reef Marine Park (GBRMP) was established through sequential declaration of sections between 1977 and 1988 with a total area of approximately 345 000 km^2. The objective was to provide for conservation of biological diversity and sustainable use of an area of recognised global environmental significance. In 1981, the Great Barrier Reef was included on the World Heritage Register. The concept of creation of significant no-take zones to address an issue of environmental primacy in a marine jurisdiction was novel and controversial. Initial zoning provided 20.6% habitat (no-trawl) protection that included NTMRs that accounted for a total of 4.6% of the entire area. The remaining 79.4% comprised General Use zones within which all forms of fishing were allowed subject to the overall provisions of the Queensland Fisheries legislation and some local MPA conditions.

In 2004, an amalgamation and revision of initial zoning and associated management came into effect addressing experience of implementation of the initial zoning, greatly increased scientific knowledge of the biodiversity and of the outstanding universal values for which the Great Barrier Reef was inscribed on the World Heritage Register. The revised zoning increased habitat protection (no-trawl) to 66% of the entire area including a greatly increased 33% of the area ascribed to implement a connective network of NTMRs. The effect of the revised zoning was that 99.5% of no-take reefs have a no-take reef within 14 km; more than 75% of fished reefs have a no-take reef within 16 km; and more than 90% have one within 22km (McCook et al., 2010).

Referring to impacts of the additional closures on areas outside them, McCook et al. (2010) show that responses differed between areas of the GBR and reflect biological and behavioural differences within and between species that are poorly understood. They note that adult coral trout (*Plectropomus* spp and *Variola* spp - main target species in the GBR) rarely move between coral reefs and that this lack of movement means that increased biomass in no-take zones will have little direct (conservation and fisheries) benefits through export of adult fish to the 2/3 of reef area that is open to fishing. In deep shoals, in southern GBR, abundance in no-take areas was twice that in fished areas and whilst some targeted species showed benefits, others showed none. In deep shoals in Central GBR, however, target fish were found to be either more abundant or less abundant, in protected shoals than in fished ones.

McCook et al. (2010) also considered that reproductive output from no-take reefs may be of enormous significance, due to disproportionately higher output per unit area from the more plentiful, larger fishes in reserves. Life cycle strategies of many targeted coral reef fish involve predictable annual short-term spawning aggregations lasting from days to weeks at specific locations. Depending on the identity of the species, the fish migrate from a few to hundreds of kilometres from their normal residential territories to spawn. Spawning aggregations can enable very high CPUE making targeted species vulnerable to over-exploitation. Spawning aggregations with consistent location and predictable timing may be protected in permanent or seasonal NTMRs under conservation legislation, by seasonal fishery closures under fishery legislation or a combination of both. Many species that exhibit such aggregations have been listed by governments as threatened or vulnerable (Gruss et al., 2014).

Elmslie et al. (2015) provide another example of inter-regional variability. They used long term data sets for 1983-2012 and 2004-2012 to compare mean density, and biomass of coral trout in the GBRMP before and after the additional closures. Density and biomass increased by 50% and 80% respectively within the closed areas. The increased body-size and abundance of coral trout within NTMRs is an important consideration for egg production and planktonic export of larvae to fished areas. Most of the targeted reef fish species are protogynous hermaphrodites with the largest females changing sex when males are removed. Larger fish have substantially higher reproductive output, larger egg size and hence likely higher larval survival. Carter et al (2017) calculated egg production per unit area (EPUA) from 2004 to 2013 for coral trout on fished and NTMR reefs throughout the GBR. They found that geographic region, NTMR status, fish size and population density all affected EPUA. The regional differences were substantial and illustrate need to understand differences of reproductive contributions of components of fish stocks, and other species, in NTMRs and throughout their range. Within-region comparison of NTMR reefs against fished reefs showed that EPUA was 21% greater in the southern region; 152% greater in the central region but 56% less in the northern region. EPUA was found to vary at both small and large spatial scales thus, despite the southern GBR having 2–4 times greater densities of *Plectopomus leopardus* than the other regions, EPUA in the southern GBR was at least one order of magnitude lower than in the central and northern GBR. This conflicts with a simple assumption that greater densities lead to greater EPUA and points to the need for better understanding and quantification of the spatial variation of EPUA to understand sink/source recruitment linkages in the design of MPA and fishery networks (Carter et al., 2017).

Finally, McCook et al (2010) they indicate that the socio-economic benefits for the main business, tourism, as well as total benefits, were very significant while those for fisheries were "*potential*" and "*concerns remain among fishers*."

Shortly after the 2004 rezoning, a Great Barrier Reef structural adjustment package for commercial fisheries was introduced with a fishery license buy-back program and revised fishery management plans. An analysis of total commercial fishery performance over the period, 2001-02 to 2013-14, reported that while the area open to forms of fishing dropped from 95.4% to 66%, fishery production dropped by 36% from 15 341 tonnes to 9 858 tonnes. The number of fishing licenses decreased by 48% and fishing effort decreased by 46%. The value of production per person-day of fishing increased by 13% over the period. These observations suggest that by restricting the area over which the fishery occurred, the catch per unit effort increased (ABS, 2015).

The outcomes of the 2004 increase in protected areas coverage have led to controversial analyses. Fletcher et al. (2015), stress that: (i) the forecast of losses was strongly underestimated (10% compared to the 30% observed); (ii) the estimated recovery time (3 years) was underestimated (more than a decade and not yet achieved). They conclude that losses are proportional to closures, allocate all the impact to these closures, and basically question their utility. However, Hughes et al. (2016), argued that (i) the future of many resources was giving serious concern before the closure; (ii) the effect reflects additional fishery management measures planned before and implemented after the; (iii) the Beyond-BACI analysis (comparing the GBR with neighbouring areas) was flawed by significant differences between the areas; and (iv) the fishery statistics used were incomplete. The lack of data on CPUEs trends outside the closed areas, for different species (e.g. predators and preys) is particularly crippling.

Altogether, the analyses by McCook et al., Fletcher et al., and Hughes et al. illustrate differences of outcomes in different areas. In the absence of an indisputable counter-factual situation, it is very difficult to empirically compare situations before and after closing an area (particularly in large and complex systems as the GBR). The positive impact inside the closure is usually confirmed even if it is variable and depends on closure duration and species. Interpretation of the impact in the fished areas remains controversial and constrained by limitations of the form and scale of data.

These observations concur with those from temperate systems which show that direct management of trawling footprints has potential to support the achievement of environmental outcomes since outside core fishing grounds, the environmental footprint of trawling on the seabed per unit of landed fish declines markedly (Jennings et al., 2012).

4.4 Closures used in demersal trawl fisheries

In this section, we use the term 'trawl' to include any form of towed fishing gear and include otter and beam trawls, seine nets and dredges. Area closures have been used as management tools in trawl fisheries for many decades. Often, they have been used to protect specific life stages (e.g. spawning areas and nursery grounds) or as effort reduction measures or as a means of reducing conflict with other fishing sectors (e.g. between mobile and static fishing gears).

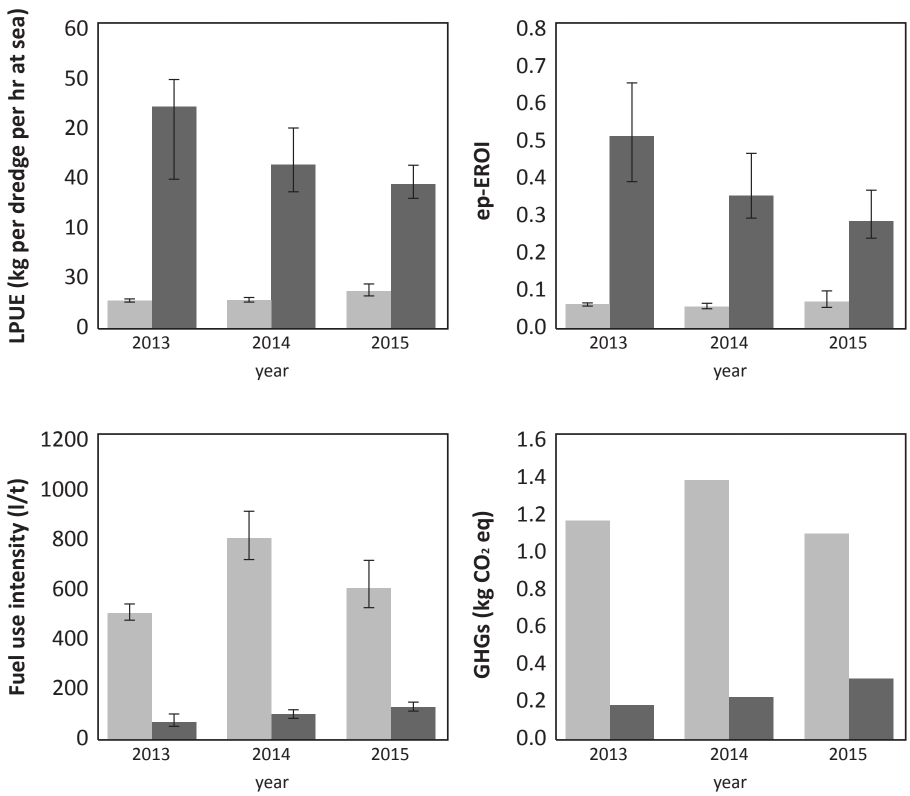
Murawski et al. (2005) reported changes in the spatial distribution of otter trawl fishing and the associated catches that occurred in response to year-round and seasonal closed areas off the coast of New England (U.S.A). Prior to the closure, 31% of trawl effort was recorded within the 22 000 km^2 of area that eventually was closed year-round. In the years following the implementation of the closure, 10% of trawl effort for the groundfish fleet was located within a 1 km wide strip adjacent to the marine protected area (MPA), with 25% of effort located within 5 km of the MPA boundary. Based on recorded CPUE data, a wide variety of groundfish (including silver hake, yellowtail flounder haddock and monkfish) showed density gradients that were consistent with (but not a confirmation of) spill-over from MPAs. These higher CPUEs were reflected in higher revenue per unit of effort within 4 km of the MPA boundary. Another unforeseen outcome of these closures was the large increase in scallop biomass within these areas which has resulted in a highly sustainable rotational scallop fishery in subsequent years.

The example above indicated a positive outcome of using large-scale spatial closures to exclude the use of towed mobile fishing gear with reference to both fish and scallops. The large scale of these area closures was probably critical to their success for the groundfish. In contrast, smaller-scale area closures have led to rapid (after 2-5 years) increases in scallop biomass and abundance (e.g. Beukers-Stewart et al., 2005; Kaiser et al., 2007; Sciberras et al., 2013; Kaiser et al., 2018). Scallops appear to be a species than can recover to high densities in a relatively short period of time provided there is an adequate supply of larvae and suitable substrata for settlement. One of the potentially important life-history bottlenecks that occur with poorly managed scallop fisheries is the secondary ecosystem habitat impact of removing settlement substratum as a result of abrasion on the seabed by the dredge gear, an effect that becomes increasingly deleterious as the spatial footprint of a fishery expands (Lambert et al. 2014). Scallops (and presumably other relatively sessile species) are well suited to spatially restricted management systems.

In the Isle of Man, the combination of a conservation zone that intentionally incorporated a designated fishing area has led to successful rebuilding of scallop biomass and has continued to sustain a regular and well-managed fishery (Bloor et al., in press). This system is relatively unusual in the northern hemisphere as it is a territorial user rights fishery (TURF) system. The system is characterised, in addition, by a high level of scientific support, detailed surveys of stock undertaken by the fishing industry, detailed knowledge of the distribution of conservation features, high compliance and effective enforcement. These factors have resulted in a fishery that is six times more profitable than open access fisheries in the surrounding territorial sea area, has minimised environment footprint to a few percent of the available area for fishing, and reduced greenhouse gas emissions through reduced fuel consumption (**Figure 1**, Bloor et al. in press).

FIGURE 1

A comparison of fisheries performance between the Ramsay Bay TURFs (black bars) and the island-wide open access fishery of Isle of Man vessels fishing for king scallops (grey bars) during a comparative time period (November and December) in the wider Isle of Man territorial sea (2013 to 2015; ICES area 37E5)



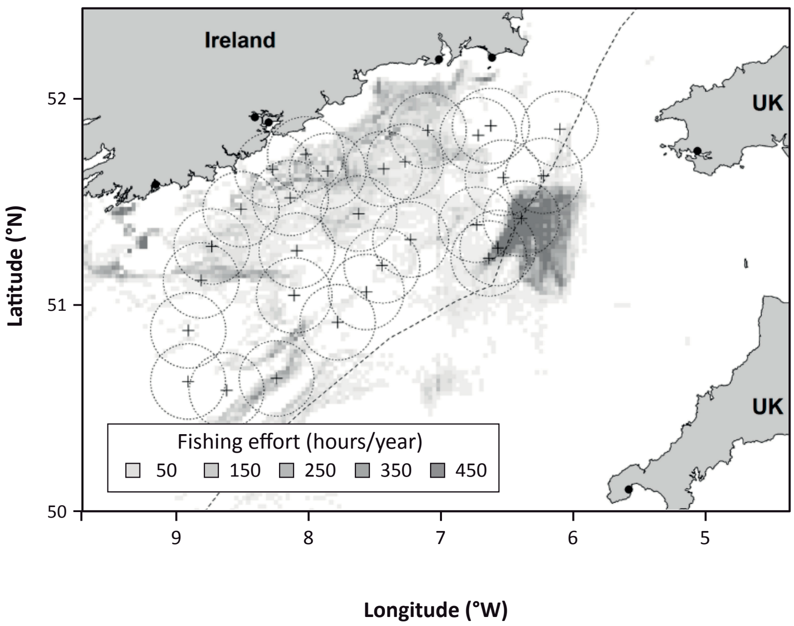
A. Catch per unit effort (kg per hour at sea per dredge); B. Fuel intensity (litres per tonne of live weight landed); C. GHG emissions (kg per kg of live weight landed); D. Edible protein – energy return on investment (ep-EROI). The 95% Confidence Intervals for both fisheries are also included on the graphs in A, B and D. Source: IFISH 2 Logbook data and Department of Environment, Forestry and Agriculture (Isle of Man Government) paper logbooks. Bloor et al. In press.

It is important to note that spatial management measures can lead to negative management outcomes for both fisheries and conservation objectives. A good illustration of this phenomenon was the unforeseen change in fishing behaviour and associated ecosystem effects in the North Sea that resulted after the implementation of the plaice box. This reserve was implemented to lower the by-catch of undersized plaice in the commercial fishery. The trawl fleet responded by ‘fishing the line’ such that vessel monitoring system records of fishing activity clearly demarcated the boundary of the plaice box with high intensities of fishing activity. As a result of the secondary effect of fishing which led to an increase in the secondary production of polychaetes (the main prey of juvenile plaice), a cultivation effect occurred whereby the by-catch of juvenile plaice increased outside the plaice box and thereby undermined its primary management objective (Hiddink et al. 2008). In a similar example, Dinmore et al. (2003) analysed the consequences of the temporary implementation of the North Sea cod box.

This closure was designed as a conservation measure to reduce effort on cod as part of a stock rebuilding programme. In subsequent assessments of the performance of the cod box the ICES advice concluded that it was not possible to determine any positive benefit for the cod stock as effort was not reduced but merely displaced into alternative areas. The displacement of activity extended the footprint of fishing on the seabed which resulted in a predicted reduction in benthic biomass (Dinmore et al. 2003).

Another often over-looked consideration is the fact that fishing activity is usually heterogeneously distributed across fishing grounds such that considerable proportions of the seabed may experience either little or no fishing which could create de facto refugia. Such areas might be considered as contributing to a conservation network. This point is nicely illustrated by an example from the Celtic Sea, in which Shephard et al. (2012) modelled elasmobranch biomass in fisheries-independent survey hauls as a function of environmental variables and fishing effort (h y^{-1}) within areas with a 20 km radius (**Figure 2**).

FIGURE 2
Distribution of fishing effort in the Celtic Sea



The heterogeneous distribution of fishing effort across fishing grounds creates 'de facto' refugia (the light-grey and white areas) for long-lived species such as skate (*Dipturus* spp.). The areas with the lowest or no fishing activity had up to 10 elasmobranch species in the fish assemblage compared with only 3 in the areas with the highest fishing intensities (Shephard et al., 2012)

Shephard et al. (2012) found that sites that occurred in the lowest 10% of the observed fishing effort range had a high number of elasmobranch species including the skate (*Dipturus* spp.). Sites with the highest fishing frequency had only three elasmobranch species. They noted that management measures that changed fisher's behaviour could displace effort into these areas and thereby eliminate their function as refugia for long-lived elasmobranch species. Interestingly, after the publication of this study, the areas with the highest occurrence of long-lived elasmobranchs were put forward by the fishing industry for formal designation as conservation zones from which fishing activities would be excluded by legislation.

4.5 Closures used in pelagic/high seas migratory fisheries

Pelagic high seas fisheries, by definition, operate in areas beyond national jurisdiction (ABNJ), and largely target stocks of highly migratory, commercially valuable pelagic species. Fisheries in ABNJ are often very remote, spread over vast areas, managed by Regional Fishery Management Organisations (RFMOs) with no central enforcement capacity, and hence often more poorly monitored than developed fisheries in EEZs. In addition, some of the target and bycatch species are poorly studied due to their highly migratory biology. Few studies have focused on the effects of spatial protection on these fish stocks.

RFMOs were introduced to address coordination of management of offshore international fisheries targeting highly migratory or straddling demersal or pelagic fish stocks (Swartz et al., 2010). They focus mainly on the management of regional fisheries on tuna and tuna-like species and are subject to multilateral treaties (e.g. the UN Fish Stocks Agreement, UNFSA) and international conventions created to promote conservation of species threatened by international trade (Convention on International Trade in Endangered Species of Wild Fauna and Flora, CITES) and highly migratory species (HMS) including species of fish, whales, seabirds and sea turtles (Convention on Migratory Species, CMS). However, overall fishing pressure on a pelagic stock is still hard to control due to the multinational nature of the fisheries and often conflicting management regimes involved. It has been estimated that about 67% of stocks managed by RFMOs are overfished or depleted (Cullis-Suzuki and Pauly, 2010).

Protection and management of HMS, biodiversity and ecosystems in ABNJ is gaining more public attention with increasing knowledge becoming available on the biology and migration behaviour of the target species and new technologies to support high seas fisheries management. Satellite-based vessel tracking through Automatic Identification System (AIS) and Vessel Monitoring Systems (VMS) that are capable of monitoring vessel location and operational status (e.g. fishing or cruising) can now aid with fleet monitoring and enforcement (McCauley et al., 2016).

While modelling and simulation studies have explored the efficacy of spatial protection for fish stock rebuilding (Gerber et al., 2003) the majority of these studies either ignores HMS or concludes that they are unlikely to benefit from spatial protection unless the protected areas are very large. Applicability of many models to the context of HMS is limited as they assume either homogenous or random distribution of fish and fishing effort (Roberts and Sargant, 2002), which does not apply to migratory fish and high seas fisheries often associated with physical (e.g. seamounts) and oceanographic features (e.g. upwellings and fronts). To improve modelling and simulation efforts, better understanding of the intrinsic linkages between a species' biology and life history, especially movement patterns, and the dynamics of associated fisheries is vital (Botsford et al., 2009). Data on this can be derived from tagging and genetic studies as well as from fishery-dependent research. RFMOs are ideally suited to collect data on fisheries associated with HMS as their convention areas cover vast regions and often multiple exclusive economic zones. Although many RFMOs collect data on the fisheries in their area, the spatial scale of reporting, aggregation of data in reported outputs and access to these data for researchers and the public can be so limited that the spatial implications for species biology and CPUE used for conventional stock assessment (ignoring most of the finer stock structures) are uninterpretable at a more detailed scale. This is particularly the case when data for individual vessels is aggregated at coarse scale on the grounds of legal or commercial sensitivity for confidentiality (see Hinz et al., 2013). Newly available studies looking at efficacy of an MPA for pelagic fisheries come from the Galápagos

Marine Reserve, Ecuador. Bucaram et al. (2017) and Boerder et al. (2017) examined the influence of the reserve on associated tuna fisheries using a mix of modelling tools, onboard observer data and publicly available satellite vessel tracking, and conclude that the reserve increased fishing productivity as well as fish stock availability. Another example of the efficacy of spatial closures for HMS comes from Baja California, Mexico. Building on previous work, Jensen et al. (2010) reported positive effects of two spatial closures in Mexican waters on the rebuilding of striped marlin (*Kajikia audax*) stocks.

Based on the scientific literature available focussing on benefits of spatial protection for HMS, three key factors appear to influence efficacy of closures for HMS and associated fisheries: (i) target species' biology, especially movement rates as well as aggregation and homing behaviour, (ii) management state of the associated fishery operating above or below maximum sustainable yield, and (iii) fishing fleet dynamics, especially attraction and displacement of fishing effort to or from closures.

Spatial protection can be adapted to species' life history traits to protect e.g. vulnerable life stages such as juveniles, and areas of importance such as feeding and spawning aggregation habitats, and fisheries need to be managed accordingly. If significant fishing effort is displaced from a protected area and not properly managed in the fishing area, it can compromise potential benefits of the closure as illustrated by the redistribution of displaced fishing effort following the closure of the Pacific High Seas Pockets and the negative impacts on bigeye tuna (*Thunnus obesus*) stocks in the Western Central Pacific (Sibert et al., 2012). This highlights the importance of a combination of spatial protection and traditional fisheries management tools such as effort and catch controls.

4.6 Contribution of fishery restricted areas (FRAs) to conservation

FRAs are fishery management areas that restrict some forms of fishing (e.g. trawling) but that permit fishing with low impact fishing techniques and can therefore contribute to conservation and biodiversity targets. A good example of this is the use of territorial user rights for fishery systems in Chile (Gelcich et al., 2005, 2008). In Chile, the Government has developed a policy to create management TURFs known as Management and Exploitation Areas for Benthic Resources (MEABRs). In these systems, cooperatives of fishermen are allocated the rights to exploit and manage specific benthic species resources within designated areas of the seabed. This leads to highly targeted and modest exploitation of a limited number of species such as the gastropod *Concholepas concholepas*, key-hole limpets *Fissurella* spp. and the red sea urchin *Loxechinus albus*. Although there are potential cascade effects that might be associated with the harvesting of these species, Gelcich et al. (2008) undertook comparative studies to understand the extent to which non-harvested (but commercially important) species may have responded within these management zones. They compared both targeted species (the focus of management) and non-targeted species that occurred in kelp forests of similar complexity within the MEABRs and in areas not subjected to this management regime. They found that populations of both the exploited and non-exploited species were enhanced within the MEABRs, compared with those areas outside, and suggested that these fishery management systems had considerable conservation benefits within the MEABR while permitting controlled exploitation. In another study, Gelcich et al. (2012) compared the performance of the MEABRs with NTMRs in the same regions and found that when compliance was good and management rules were well enforced within the MEABRs and the NTMRs their biological characteristics were not different. However, in those MEABRs where compliance and enforcement were weak, the densities of commercially important species were significantly lower than in NTMRs.

5. DISCUSSION

The performance of fishery and biodiversity management systems depends on the wellbeing of marine environment. A human-induced degradation or a natural oscillation of the marine system may impede even draconian attempts to reverse trends in species depletion (e.g. cod in Newfoundland remain under very strict management conditions imposed for several decades). For both management systems, the primary objective of one has potential benefits and/or costs for that of the other. The SDG 14 goals for fisheries and biodiversity make no reference to the form, precedence or precautionary arrangements for inter-sectoral management of risks reflected in the expressed expectations of current and future access to fisheries and maintenance of marine biodiversity and ecosystem processes. Rice et al (2012) reported the outcomes of a workshop of fisheries and conservation specialists considering the role of MPAs in fisheries management. An expert opinion process was used to score objectives of fishery management and biodiversity conservation interests drawing on the experiences of workshop participants. Whilst the workshop was not broadly representative, it indicated that half of all fisheries objectives and 40% of biodiversity objectives were considered likely to receive support from both perspectives. Conversely, only 25% of fishery objectives and 30% of conservation objectives were likely to be the cause of fishery/conservation conflict.

There are tensions between fish stock and biological diversity conservation management systems that should be explicitly addressed. Determining the effect of a well-managed NTMR on a stock or set of stocks is particularly difficult because it depends substantially on the adequacy of information from substantial socio-economic and biological research and knowledge sharing. It requires data on species and community biology, past, current and expected fishing effort, socio-economic dependency and compliance with protected area regulations and with fishing regulations in the fished stock area. Indeed, for many developing nations and particularly for coastal, fisheries we lack robust data on recruitment, natural mortality, food web linkages, fish catch, fishing effort, compliance, local and market dependencies, economics and other human impacts on stocks.

The examples addressed in this chapter demonstrate that effective implementation of removal of fishing activities from defined areas of the sea most often results in an increase in abundance, biomass, diversity and also reproductive output in the areas from which the fishing activities are removed. In addition, areas with partial protection can generate similar responses, but these responses are generally less pronounced than the response seen within NTMRs or areas from which all extractive activities are removed (Sciberras et al., 2015). Depending on the biology and range of target species, a NTMR may have positive effects in stock maintenance or rebuilding for some species but no discernible effect for other species with large ranges or recruitment linkages (Claudet et al., 2010). There is some evidence to support the observation of spillover from some NTMRs with declining CPUE of quasi resident species as fisheries move further away from the boundary of NTMRs (Abesamis at al., 2006; Lester et al., 2009; Murawski et al., 2005; Russ et al., 2015). Most of these studies relate to nearshore areas and target species with limited or intermediate range and represent situations with a reasonably high level of support for and compliance with fishery constraints.

Adequate monitoring programmes may be able to demonstrate increased biomass of species within a well-managed NTMR which may be accompanied by increased stock biomass, stable stock biomass or falling stock biomass for the same species outside of the NTMR. Where beneficial responses that result from NTMRs and partial protection areas have occurred, typically they take more than 3 – 5 years to occur (e.g. Russ et al., 2004; Sciberras et al., 2013; Murawski et al., 2000). This presents challenges of managing displaced or increasing fishing effort in the fished areas outside the spatially protected area.

The introduction of a NTMR results in an extended burden for the displaced fishers until any benefits from the NTMR become apparent outside the boundaries. This may in turn result in increased effort through displacement into other areas unless a reduction in effort is implemented (as in the GBR). This medium-term shadow of lost fishing opportunities was elegantly demonstrated by Steele and Beet (2003) who compared the performance of conventional fishery management tools (e.g. effort reduction) and the use of NTMRs as stock-rebuilding tools. Fishing effort reductions resulted in instantaneous positive effects on stock status and had a modest impact on loss of fishing opportunities, whereas the use of NTMRs led to an instantaneous substantial reduction in fishery yield which did not recover to a point that compensated for the initial loss in fishing opportunities until 5 years after implementation. The timescale for compensation to occur when using an NTMR as a fishery management tool is similar to the response time-scales for species recovery within NTMRs (e.g. Murawski et al., 2000). While some site-specific conservation features or species might need instantaneous protection by excluding certain fishing activities in some areas, fisheries rebuilding can also be undertaken at a slower pace which avoids a catastrophic loss of livelihoods (Steele and Beet, 2003).

At present, we consider that, for most fisheries, the data are inadequate for anything other than speculation about or modeling the response of a 'stock' or population to the use of spatial closures such as NTMRs or partial protection areas. Current monitoring and management of fisheries is undertaken at large spatial scales over 1000s km^2 although greater management precision is possible where VMS and AIS data are made available so that landings or catch records can be more precisely resolved with respect to catch location. While NTMRs and different fishery stocks may be encompassed within such areas, the design of monitoring programs is focused on measuring overall trends in the change in population status in a consistent manner. These programmes primarily address recruitment and fishing effort/mortality in the system and occasionally take account of limited environmental parameters such as temperature. While fishery data may be collected at finer scales, it may be aggregated and legally protected on grounds of commercial confidentiality. This can constrain provision of data at a spatial or temporal scale that would enable the definitive establishment of a causal spatial relationship between a change in recruitment or mortality and other impacts within the geographic range of a stock. The determination of such links requires a more experimental approach with surveys designed to answer the specific question about whether a NTMR is the causal factor for a change in stock status. This inability to differentiate the effects of an NTMR on stock status is further compounded by the additional management measures that are often applied in conjunction with the use of the NTMR. Only in the absence of change in all other management measures or in climatic conditions would it be possible to infer that any positive change in the status of a stock was directly attributable to the implementation of an NTMR. Nevertheless, it is worth remembering that, in many areas of the world, particularly but not only in small-scale fisheries, even the most basic information on catches and landings is lacking and hence the possibility to precisely determine such effects is extremely limited. To this extent it is probably impossible to break away from the use of expert opinion as the likely consequences of the effect of the implementation of a NTMR, facing nonetheless the possibility of happy or nasty "surprises".

Considerations of the use of spatial management measures to enhance stock status aside, areas may also be permanently closed to fishing for the purposes of biodiversity conservation, scientific research, or safety and security of military, navigational or industrial facilities (e.g. windfarms). Within such areas it is important to understand and monitor both biodiversity and potential responses in fish and shellfish stocks (e.g. Gelcich et al., 2012). When areas are closed (or planning to be closed), the loss of fishing incomes and the shadow effect identified by Steele & Beet (2003) necessarily becomes an important consideration to achieve buy-in by stakeholders and improve chances of success. In such cases a positive change in biomass, age and size structure will presumably increase resilience of surrounding stocks.

An important factor in the history of the use of NTMRs is the engagement of the local governance arrangements, fisher and other marine user communities as primary stakeholders in the process of development (Gelcich and Donlan 2015). This requires a process of knowledge-sharing and building trust about managing marine resources, understanding of basic marine ecological concepts, community organisation and empowerment. In some cases, fishers come to perceive the benefits of the use of NTMRs after a prolonged period during which the biological benefits of such areas become apparent (Steele and Beet 2003; Blyth-Skyrme et al. 2004; Gelcich et al. 2008). If some fishers are lost from the system due to reduced fishing opportunities, the remaining fishers are likely to achieve greater harvests and profitability which may underpin misleading change towards a positive attitude to effective spatial management systems.

Areas may be closed to fishing in order to address objectives of biodiversity conservation or fishery management. A closed area may contribute to the achievement of either or both objectives but it is important to avoid ungrounded expectation of benefits against one objective from a closure to address the other. Scientists and managers need to ensure that they do not 'promise' stock rebuilding benefits from NTMRs if these are uncertain. Unrealised benefits harm relationships with stakeholders and undermine future relationships that depend on trust (Jones 2012). One of the case-studies that sheds some light on the importance of this issue is the Ramsay Bay fishery in the Isle of Man. In this fishery, both fishermen and Government agencies realised the need to close the area to fishing for a period of time to enable recovery of the scallop stock. This provided the platform to devise a novel management area that integrated partial protection areas with conservation objectives alongside a fishery management area that was leased to the producer organisation. The outcome of introducing this TURF (which ensured future resources ownership) was greater fishing profitability when compared with the surrounding open access fishery. Nevertheless, even this fishery may have unknown spillover effects (benefits) for the wider open access fishery but these benefits (if they occur) are difficult to assess.

To conclude, while NTMRs and partial exclusion zones often are associated with positive changes in community metrics within their boundaries and have been observed to generate near-field spillover effects, the potential benefits of these systems remain difficult to quantify at the level of a 'stock' or population, as indeed at the level of the ecosystem in relation to biodiversity conservation. This difficulty arises from a mismatch in the scale of current monitoring programmes that were not resourced or designed to specifically address the question of the extent to which NTMRs contribute to improve stock status. The situation is one in which proof is hard to establish and absence of proof of effect is not proof of absence of effect. This prognosis may seem pessimistic but could be changed quite simply if there was greater determination to understand more clearly the effect of using NTMRs on wider stock status. This could be achieved with a proper time-limited experimental approach that examined multiple lines of evidence that include: stock metrics within and beyond NTMRs, reproductive output, recruitment, environmental variation within the timescale of the experiment, and social and economic indicators of performance and well-being. The latter approach would be facilitated by greater integration between the fisheries and conservation science disciplines. Scientists working in these respective fields overlap considerably in expertise, but the policy questions that they are tasked with answering do not facilitate the consideration of questions that span these discipline areas.

6. REFERENCES

Abesamis, R.A., Alcala, A.C. & Russ, Garry, R. 2006. How much does the fishery at Apo Island benefit from spillover of adult fish from the adjacent marine reserve? *Fishery Bulletin*, 104 (3). pp. 360-375.

ABS. 2015. Information Paper: An Experimental Ecosystem Account for the Great Barrier Reef Region. Australian Bureau of Statistics. Accessed at http://www.abs.gov.au/ausstats/abs@.nsf/Latestproducts/4680.0.55.001Main%20Features302015?opendocument&tabname=Summary&prodno=4680.0.55.001&issue=2015&num=&view= 28 November 2017.

Alcala, A.C. & Russ, G.R. 2006. No-take marine reserves and reef fisheries management in the Philippines: a new people power revolution. *AMBIO: A Journal of the Human Environment*, 35(5): 245-254.

Blyth-Skyrme, R.E., Kaiser, M.J., Hiddink, J.G,, Edwards-Jones, G. & Hart, P.J. 2006. Conservation benefits of temperate marine protected areas: variation among fish species. *Conservation Biology*, 20(3): 811-820.

Beukers-Stewart, B.D., Vause, B.J., Mosley, M.W., Rossetti, H.L. & Brand, A.R. 2006. Closed areas and stock enhancement of scallops: what's the catch. *Journal on Shellfish Research*, 25: 267-268.

Bloor, I., Dignan, S., Emmerson, J., Beard, D., Gell, F.E., Duncan, P., Kennington K. et al. In press. Territorial User Rights Fishery (TURF) system produces economic and conservation dividends. *Conservation Letters*, 00: 00-00

Boerder, K., Bryndum-Buchholz, A. & Worm, B. 2017. Interactions of Tuna Fisheries with the Galápagos Marine Reserve. *Marine Ecology Progress Series*, 585: 1-15

Botsford, L. W., Brumbaugh, D. R., Grimes, C., Kellner, J. B., Largier, J., O'Farrell, M. R., Ralston, S. et al. 2009. Connectivity, sustainability, and yield: bridging the gap between conventional fisheries management and marine protected areas. *Reviews in Fish Biology and Fisheries*, 19(1): 69-95.

Bucaram, S. J., Hearn, A., Trujillo, A. M., Rentería, W., Bustamante, R. H., Morán, G., Reck, G., et al. 2018. Assessing fishing effects inside and outside an MPA: The impact of the Galapagos Marine Reserve on the Industrial pelagic tuna fisheries during the first decade of operation. *Marine Policy*, 87, 212-225.

Carter, A.B., Davies, C.R., Emslie, M.J., Mapstone, B.D., Russ, G.R., Tobin, A.J. & Williams, A.J. 2017. Reproductive benefits of no-take marine reserves vary with region for an exploited coral reef fish. *Scientific reports*, *7*(1): 1-12

CBD. 2004. COP VII Decision 5. Marine and Coastal Biodiversity. Accessed 05 February 2018 at https://www.cbd.int/doc/meetings/sbstta/sbstta-10/official/sbstta-10-08-add1-en.pdf

CBD. 2011. Aichi Biodiversity Targets. https://www.cbd.int/sp/targets/default.shtml

Claudet, J., Osenberg, C.W., Benedetti-Cecchi, L., Domenici, P., García-Charton, J.A., Pérez-Ruzafa, Á., Badalamenti, F., et al. 2008. Marine reserves: size and age do matter. *Ecology letters*, 11(5): 481-489.

Claudet, J., Osenberg, C.W., Domenici, P., Badalamenti, F., Milazzo, M., Falcón, J.M Bertocci, I. et al. 2010. Marine reserves: fish life history and ecological traits matter. *Ecological applications*, 20(3): 830-839.

Cudney-Bueno, R., Lavín, M.F., Marinone, S.G., Raimondi, P.T. & Shaw, W.W. 2009. Rapid effects of marine reserves via larval dispersal. *PLOS One*, 4(1): e4140.

Cullis-Suzuki, S. & Pauly, D. 2010. Failing the high seas: a global evaluation of regional fisheries management organizations. *Marine Policy*, 34(5): 1036-1042.

Day, J., Dudley, N., Hockings, M., Holmes, G., Laffoley, D.D.A., Stolton, S. & Wells, S.M. 2012. *Guidelines for applying the IUCN protected area management categories to marine protected areas*. Gland (Switzerland). IUCN: 36 p.

Dinmore, T.A., Duplisea, D.E., Rackham, B.D., Maxwell, D.L. & Jennings, S. 2003. Impact of a large-scale area closure on patterns of fishing disturbance and the consequences for benthic communities. *ICES Journal of Marine Science*, 60(2): 371-380.

Dobbs, K., Day, J., Skeat, H., Baldwin, J., Molloy, F., McCook, L., Johnson, M. et al. 2011. Developing a long-term outlook for the Great Barrier Reef, Australia: a framework for adaptive management reporting underpinning an ecosystem-based management approach. *Marine Policy,* 35(2) : 233-240.

Dudley, N., ed. 2008. *Guidelines for Applying Protected Area Management Categories*. Gland, Switzerland: IUCN. x + 86pp.

Ecoutin, J-M., Simier, M., Albaret J-J., Laé, R., Raffray, J., Sadio, O. & Tito de Morais, L. 2014. Ecological field experiment of short-term effects of fishing ban on fish assemblages in a tropical estuarine MPA. *Ocean & Coastal Management*, 100: 74-85

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

Emslie, M.J., Logan, M., Williamson, D.H., Ayling, A.M., MacNeil, M.A., Ceccarelli, D., Cheal, A.J. et al. 2015. Expectations and outcomes of reserve network performance following re-zoning of the Great Barrier Reef Marine Park. *Current Biology*, 25(8): 983-992.

FAO. 2016. *The State of World Fisheries and Aquaculture 2016. Contributing to food security and nutrition for all*. Rome: 200 p.

Fletcher, W. J., Kearney, R. E., Wise, B. S. & Nash, W. J. 2015. Large-scale expansion of no-take closures within the Great Barrier Reef has not enhanced fishery production. *Ecological Applications*, 25(5):1187-1196.

Gelcich, S., Edwards-Jones, G., Kaiser, M.J. & Watson, E., 2005. Using discourses for policy evaluation: the case of marine common property rights in Chile. *Society and Natural Resources*, 18(4): 377-391.

Gelcich, S., Kaiser, M.J., Castilla, J.C. & Edwards-Jones, G. 2008. Engagement in co-management of marine benthic resources influences environmental perceptions of artisanal fishers. *Environmental Conservation*, 35(1): 36-45.

Gelcich, S., Fernández, M., Godoy, N., Canepa, A., Prado, L. & Castilla, J.C. 2012. Territorial user rights for fisheries as ancillary instruments for marine coastal conservation in Chile. *Conservation Biology*, 26(6), pp.1005-1015.

Gelcich, S. & Donlan, C.J. 2015. Incentivizing biodiversity conservation in artisanal fishing communities through territorial user rights and business model innovation. *Conservation Biology*, 29(4): 1076-1085.

Gerber, L. R., Botsford, L. W., Hastings, A., Possingham, H. P., Gaines, S. D., Palumbi, S. R., & Andelman, S. 2003. Population models for marine reserve design: a retrospective and prospective synthesis. *Ecological Applications*, S47-S64.

Guidetti, P., Milazzo, M., Bussotti, S., Molinari, A., Murenu, M., Pais, A., Spano, N., et al. 2008. Italian marine reserve effectiveness: does enforcement matter? *Biological Conservation*, 141(3): 699-709.

Guidetti, P. & Claudet, J. 2010. Co-management practices enhance fisheries in marine protected areas. *Conservation Biology*, 24(1): 312-318

Grüss, A., Karnauskas, M., Sagarese, S.R., Paris, C.B., Zapfe, G., Walter, J.F., Ingram, W. & Schirripa, M.J., 2014. Use of the Connectivity Modeling System to estimate the larval dispersal, settlement patterns and annual recruitment anomalies due to oceanographic factors of red grouper (*Epinephelus morio*) on the West Florida Shelf. SEDAR42-DW-03, SEDAR, North Charleston, SC.

Halpern, B.S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? *Ecological applications*, S117-S137.

Hiddink, J.G., Rijnsdorp, A.D. & Piet, G. 2008. Can bottom trawling disturbance increase food production for a commercial fish species? *Canadian Journal of Fisheries and Aquatic Sciences*, 65(7): 1393-1401.

Hinz, H., Murray, L.G., Lambert, G.I., Hiddink, J.G. & Kaiser, M.J. 2013. Confidentiality over fishing effort data threatens science and management progress. *Fish and Fisheries*, 14(1): 110-117.

Hughes, T. P., Cameron, D.S., Chin, A., Connolly, S.R., Day, J.C., Jones, G.P., McCook, L. et al. 2016. A critique of claims for negative impacts of Marine Protected Areas on fisheries. *Ecological Applications*, 26(2): 637–641.

Jennings S., Kaiser M.J. & Reynolds J.D. 2001. *Marine Fisheries Ecology*. Blackwell Science, Oxford: 432 p.

Jennings, S, Lee, J. & Hiddink, J.G. 2012. Assessing fishery footprints and the trade-offs between landings value, habitat sensitivity, and fishing impacts to inform marine spatial planning and an ecosystem approach. *ICES Journal of Marine Science*, 69(6): 1053-1063.

Jensen, O. P., Ortega-Garcia, S., Martell, S. J., Ahrens, R. N., Domeier, M. L., Walters, C. J., & Kitchell, J. F. 2010. Local management of a "highly migratory species": the effects of long-line closures and recreational catch-and-release for Baja California striped marlin fisheries. *Progress in Oceanography*, 86(1), 176-186

Jones, P.J. 2012. Marine protected areas in the UK: challenges in combining top-down and bottom-up approaches to governance. *Environmental Conservation*, 39(3): 248-258.

Johannes, R.E. 1981. Working with fishermen to improve coastal tropical fisheries and resource management. *Bulletin of Marine Science*, 31(3): 673-680.

Kaiser M.J., Blyth-Skryme R.E., Hart P.J.B., Edwards-Jones G. & Palmer D. 2007. Evidence for greater reproductive output per unit area in areas protected from fishing. *Canadian Journal of Fisheries & Aquatic Science*, 64: 1284-1289.

Kaiser M.J., Hormbrey S., Booth J.R., Hinz H. & Hiddink J.G. 2018. Recovery linked to life-history of sessile epifauna following exclusion of towed-mobile fishing gear. *Journal of Applied Ecology,* 55(3), 1060-1070.

Kenchington, R. & Hutchings P. 2012. Science, biodiversity and Australian management of marine ecosystems. *Ocean and Coastal Management*, 69, 194-199

Kerwath, S.E., Winker, H., Götz, A. & Attwood, C.G. 2013. Marine protected area improves yield without disadvantaging fishers. *Nature Communications*, 4 : 2347.

Lambert G.I., Jennings S., Kaiser M.J., Davies T.W. & Hiddink J.G. 2014. Quantifying recovery rates and resilience of seabed habitats impacted by bottom fishing. *Journal of Applied Ecology,* 51: 1326-1336

Lester, S.E., Halpern, B.S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B.I., Gaines, S.D., et al. 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series*, 384: 33-46.

McCauley, D. J., Woods, P., Sullivan, B., Bergman, B., Jablonicky, C., Roan, A., Hirschfield, M. et al. 2016. Ending hide and seek at sea. *Science*, 351(6278): 1148-1150.

McClanahan, T.R. & Mangi, S., 2000. Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. *Ecological applications*, 10(6): 1792-1805.

McCook, L.J., Ayling, T., Cappo, M., Choat, J.H., Evans, R.D., De Freitas, D.M., Heupel, M. et al. 2010. Adaptive management of the Great Barrier Reef: a globally significant demonstration of the benefits of networks of marine reserves. *Proceedings of the National Academy of Sciences*, 107(43), pp.18278-18285.

Murawski, S.A., Brown, R., Lai, H.L., Rago, P.J. & Hendrickson, L. 2000. Large-scale closed areas as a fishery-management tool in temperate marine systems: the Georges Bank experience. *Bulletin of Marine Science*, 66(3): 775-798.

Murawski, S.A., Wigley, S.E., Fogarty, M.J., Rago, P.J. & Mountain, D.G. 2005. Effort distribution and catch patterns adjacent to temperate MPAs. *ICES Journal of Marine Science*, 62(6): 1150-1167.

Papworth, S.K., Rist, J., Coad, L. & Milner-Gulland, E.J. 2009. Evidence for shifting baseline syndrome in conservation. *Conservation Letters*, 2(2): 93-100.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. *Trends in ecology and evolution*, 10(10): 430.

Rice, J., Moksnsess, E., Attwood. C., Brown, S.K., Dahle, G., Gjerde, K.H., Grefsund, E.S. et al. 2012. The role of MPAs in reconciling fisheries management with conservation of biological diversity. *Ocean and Coastal Management*, 69: 217-230.

Roberts, C. M. & Sargant, H. 2002. Fishery benefits of fully protected marine reserves: why habitat and behavior are important. *Natural Resource Modeling*, 15(4), 487-507.

Ruddle, K. 1998. Traditional community-based coastal marine fisheries management in Viet Nam. *Ocean & Coastal Management*, 40(1):1-22.

Russ, G.R. 2002. Yet another review of marine reserves as reef fishery management tools. *In:* Sale, P.S., *ed. Coral reef fishes: dynamics and diversity in a complex ecosystem*. Elsevier, San Diego, CA, USA: 421-443.

Russ, G.R., Alcala, A.C., Maypa, A.P., Calumpong, H.P. & White, A.T. 2004. Marine reserve benefits local fisheries. *Ecological applications*, 14(2): 597-606.

Russ, R., Miller, K. I., Rizzari, J. R, & Alcala, A. C. 2015. Long-term no-take marine reserve and benthic habitat effects on coral reef fishes. *Marine Ecology Progress Series*, 529: 233-248.

Sale, P.F., Agardy, T., Ainsworth, C.H., Feist, B.E., Bell, J.D., Christie, P., Hoegh-Guldberg, O., et al. 2014. Transforming management of tropical coastal seas to cope with challenges of the 21st century. *Marine Pollution Bulletin*, 85(1): 8-23.

Sciberras M., Hinz H., Bennell J., Jenkins S.R., Hawkins S.J. & Kaiser M.J. 2013. Benthic community response to a scallop closure in a dynamic seabed habitat. *Marine Ecology Progress Series*, 480: 83-98.

Sciberras M., Jenkins S.R., Mant R., Kaiser M.J., Hawkins S.J. & Pullin A.S. 2015. Evaluation the relative conservation value of fully and partially protected marine areas. *Fish and Fisheries*, 16: 58-77.

Shephard, S., Gerritsen, H., Kaiser, M.J. & Reid, D.G. 2012. Spatial Heterogeneity in Fishing Creates *de facto* Refugia for Endangered Celtic Sea Elasmobranchs. *PLOS One*, 7(11): e49307. https://doi.org/10.1371/journal.pone.0049307

Sibert, J., Senina, I., Lehodey, P. & Hampton, J. (2012). Shifting from marine reserves to maritime zoning for conservation of Pacific bigeye tuna (*Thunnus obesus*). *Proceedings of the National Academy of Sciences*, *109*(44), 18221-18225.

Steele, J.H. & Beet, A.R. 2003. Marine protected areas in non-linear ecosystems. *Proceedings of the Royal Society of London B: Biological Sciences*, 270(Suppl 2): S230-S233.

Stephenson, W., Williams, W.T. & Lance, G.N. 1970. The macrobenthos of Moreton Bay. Ecol. Mon, 40: 459–494

Stewart, G.B., Kaiser, M.J., Côté, I.M., Halpern, B.S., Lester, S.E., Bayliss, H.R. & Pullin, A.S. 2009. Temperate marine reserves: global ecological effects and guidelines for future networks. Conservation Letters, 2(6): 243-253.

Swartz, W., Sala, E., Tracey, S., Watson, R. & Pauly, D. 2010. The spatial expansion and ecological footprint of fisheries (1950 to present). *PLOS One*, 5(12): e15143.

United Nations. 2016 *Sustainable Development Goal 14: Conserve and sustainably use the oceans, seas and marine resources*. Accessed 5/02/2018 at http://www.un.org/sustainabledevelopment/oceans/

Woodcock P., O'Leary B.C., Kaiser M.J. & Pullin A.S. 2017. Your evidence or mine? Systematic evaluation of reviews of marine protected area effectiveness. *Fish and Fisheries*, 18: 668-681. http://dx.doi.org/10.1111/faf.12196

Worm, B., Barbier, E.B., Beaumont, N., Duffy, J.E., Folke, C., Halpern, B.S., Jackson, J.B., et al. 2006. Impacts of biodiversity loss on ocean ecosystem services. *Science*, 314(5800): 787-790.