An Analysis of the Impact on Māori Property Rights

in Fisheries of Marine Protected Areas and Recreational Fishing Outside the Quota Management System

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This project was commissioned to provide an independent analysis of ecosystem-based management (EBM) and marine protected area (MPA) application in New Zealand and the consequence of reallocation of fishing rights to recreational fishing, specifically focusing on the implications that these approaches have for Māori rights in fisheries. As the expansion of MPAs in New Zealand follows from international efforts to establish or expand MPAs and apply the concept of EBM more generally, this project begins by carrying out a critical overview of the adoption of these approaches worldwide. This international context is then used to examine the New Zealand experience and implications for Māori fisheries rights specifically.

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Foreword

A key expectation of the Deed of Settlement was to enable iwi to regain tino rangatiratanga over the access to, and use of, fisheries resources. Māori accepted the fishing rights they secured under the Fisheries Deed of Settlement are subject to a responsibility to ensure sustainability.

The acceptance of the Quota Management System (QMS) as a means of utilising marine resources for commercial purposes was founded on the expectation that the rights allocated under the QMS were secure.

However, despite the success of the QMS in reversing the decline of our inshore fisheries and in providing a framework for the development of deepwater fisheries, areas where we go fishing are at risk of being closed to fishing. Further, in some cases we are seeing proportional reductions in the access we have to sustainably available fisheries because of the lack of enforcement of recreational allowances.

We searched the world to find a suitably qualified person to investigate the impact that closing off areas to fishing could have on our rights guaranteed under the Deed of Settlement, along with the impact of managing recreational fishing outside the QMS.

Professor Gary Libecap is renowned for his work in environmental economics and Mike Arbuckle has extensive experience with the management of fisheries in Aotearoa New Zealand, including as a lead author in the review of the performance of the QMS during its first thirty years. Gary and Mike were ably assisted by Chester Lindley as the lead researcher for this work.

Are marine protected areas actually a successful tool for protecting ecosystems from risks? If they are indeed a successful way of safeguarding our ecosystems, we would need to incorporate this into our policy advice and discussions with iwi. If not, we need to ask: Why are marine protected areas still proposed as the primary response to ecosystem protection when the adverse impact on Māori rights, local communities and even the long-term effectiveness of alternative responses appears to outweigh the benefits?

The idea of 'protected areas' in the marine environment stems from the Convention on Biological Diversity. In Aotearoa, we are grappling with marine protected areas as a management tool: What are they and what problems are they intended to address?

These questions are central to the government's work on a new biodiversity strategy, its review of marine protected areas and its position on a new global framework for managing biodiversity, both within national jurisdictions and on the high seas. It is also central to its proposals to establish marine protected areas within several regions around the country.

In this report, Professor Libecap and his team provide insights into the problems of implementing marine protected areas. They uncover vague problem definition and goals for marine protected areas, lack of agreed criteria for their establishment, lack of cost-benefit analysis and lack of integration with national laws and indigenous rights. They observe that local, rights-based systems result in more positive incentives for resource users to manage the effects of fishing on fisheries and the marine environment.

On the issue of reallocation, the team finds that reallocating quota in shared fisheries to a less regulated recreational sector contradicts the objectives of the Deed of Settlement and the spirit of collaborative management of marine resources.

I commend this report as a major contribution to the discussions necessary to ensure marine policy supports our ongoing relationship with Tangaroa and ensures the Deed of Settlement endures.

Rangimarie Hunia
Chair – Te Ohu Kaimoana
March 2020

Project Overview

New Zealand Fishery Regulations

New Zealand is considered a leader in fisheries management under a property-rights system that is unusual worldwide in the security, durability, and definition of the rights held (Hale and Rude, 2017). Although rights-based management has been shown to bring remarkable benefits by changing incentives regarding exploitation (Grafton et al., 2000; Costello et al., 2008) in most settings elsewhere in the world, the property right is less well defined and secure (Grainger and Costello, 2014). New Zealand's fishery policies are based on the Fisheries Amendment Act 1986 that implemented the quota management system (QMS) for twenty-nine species (Newell et al., 2005; Day, 2004,) and the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992, which formally recognised Māori customary, non-commercial fishing rights and rights to manage their fisheries, and guaranteed Māori 10 percent of the quota for existing commercial fisheries placed in the QMS, 20 percent of any new fisheries brought subsequently into the QMS, and 50 percent of Sealord Products Limited.

In this regard, New Zealand is in the forefront of the protections and property rights granted to its indigenous population, Māori. The Deed of Settlement 1992 and related legislation not only provides formal property rights but recognises the customary fishing rights and management practices of Māori due to their long-standing and cultural participation in the fishery (Day, 2004). Elsewhere in the world, indigenous populations do not have such property rights or the protections called for under the United Nations Declaration on the Rights of Indigenous Peoples (UNDRIP) agreed to by 144 nations in 2007. In Latin America and Asia, indigenous populations typically are not recognised formally by governments as having property rights to the critical customary resources that they use and that have cultural significance. In many cases, exploitation rights are granted to others without the approval of indigenous populations, and these rights also deny the indigenous people's access to and the ability to benefit from them (United Nations Permanent Forum on Indigenous Issues, n.d.). In the United States and Canada, property rights to resources are held in trust by national governments and are not defined by or delegated to indigenous groups. Without property rights to essential resources and dependence on remote bureaucratic administrative agencies for use, income, and social services, vital traditions and customary practices have deteriorated. By any socio-economic measure, indigenous populations in the United States and Canada perform poorly compared with the rest of the population.

The property rights held by Māori under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992 that make them unique, as well as the rights held by other fishers within the QMS, are at risk from:

1. Proposed expansion of marine protected areas (MPAs), beyond those under the Marine Reserves Act 1971, in order to provide ecosystem-based fishery management (EBFM) (Ministry for the Environment, 2016). These actions involve not only large MPAs and commercial fishing exclusion zones around the Kermadec Islands, Campbell Islands, parts of the Hauraki Gulf, and other areas under discussion as part of the South East Marine Protection Forum but also introduce a major change in New Zealand's approach to fisheries management and marine protection. These large spatial reserve areas are installed unilaterally, undermining property rights to existing and new fisheries as well as customary access and management by Māori. Such MPA expansions are potential takings and inconsistent with provisions of the Deed of Settlement 1992. Moreover, their implementation does not recognise the actions taken by Māori or other QMS quota holders to safeguard the marine environment.

2. Gradual, effective reduction of Māori quota within the southern bluefin tuna quota management system through reallocation to the unregulated sports fishing industry. As above, there are also negative impacts on other QMS quota holders as well as the QMS overall.

Marine Protected Areas Expansion

Proposed expansion of marine protected areas in New Zealand follows from international efforts to establish or expand MPAs in order to achieve EBFM. This action is despite the fact that arrangements are currently in place in New Zealand to account for ecosystem-based management (Hale and Rude, 2017). The proposed MPA extensions are spearheaded by non-government organisations, international agencies, consultants, and delegations from administrative agencies within signatory governments. International efforts include provisions of the United Nations Framework Convention on Climate Change whereby member nations committed to designate new marine protected areas by 2009; the 1992 United Nations Conference on Environment and Development (UNCED) and the Convention on Biological Diversity (CBD) that encouraged the use of protected areas or area-based closures; and the 2002 World Summit on Sustainable Development (WSSD), where MPAs were placed at the top of the agenda for implementation in order to achieve sustainability (Wells et al., 2008).

MPAs often involve vast areas of sea being set aside, effectively placing them off-limits to fishers and other resource users and for other applications. MPAs are put under centralised, bureaucratic monitoring and management by national and international agencies. The officials within these agencies are not elected, nor are they generally directly accountable to the citizens of the countries whose waters are affected. Typically, in administering their mandates, they are driven by biological objectives, without weighing up trade-offs or considering area-specific institutions or fishery management practices. Advocates and bureaucratic officials can work together because new regulatory mandates around establishing MPAs and EBFM benefit both the agencies and advocacy groups. They are not disinterested parties, and regulatory officials, often who have tenure, do not bear the direct costs of their actions. This setting creates an inherent bias towards implementation of EBFM and MPAs unless there is a strong, well-organised competitive interest group that opposes such actions (Becker, 1983). As described below, such groups are absent in most world fisheries where MPAs have been implemented or are under consideration. Because of their property rights, Māori are an exception who could both protect their rights under the Deed of Settlement 1992 and force better articulation and defence against proposals to expand MPAs.

Phrasing the creation or extension of MPAs as EBFM cloaks these regulatory impoundments as essential for provision of public goods. While it is true that traditional fisheries management, focused on single species sustainability, may ignore some ecosystem considerations, it is not necessarily the case that fishers neglect such issues because they can be directly affected. Their incentives to consider broad habitat effects depend on the nature of the property rights they hold – their strength and durability. Because of their property rights under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992, Māori not only have incentives but the ability to address issues such as bycatch and gear impacts on habitat without unilateral declaration of the need for EBFM. Their cultural and community values along with the value of quota under the QMS depend on vibrant fish stocks and healthy ecosystems. These positive incentives also generally apply to other quota holders within the QMS. If local institutions are undercut and existing fishers displaced, how might incentives and practices change and thwart achievement of the ecosystem goal?

In other parts of the world where fishing rights are non-existent (open access) or weak or fishery access and management are plagued by corruption, there may be fewer direct incentives to protect stock and ecosystems. In those cases, any remedy would most directly lie in more precise definition and strength of the property right rather than in imposing broad, spatial MPAs that overlay and displace local users with bureaucratic and political administrators. Further, with an absence of strong national institutions, it seems unlikely that exogenously-defined MPAs will achieve their goals, given the need for effective monitoring and enforcement.

Unfortunately, there is little precision in the definition of EBFM or consensus on what the concept means, when it should be implemented, where it should be applied, or what it is to deliver. A survey of experts in fisheries science and management reveals a wide variety of opinions and lack of consensus on which actions may be part of ecosystem-based management (Trochta et al., 2018). These results are not surprising in a setting where advocates and bureaucratic officials are not required to use general, consensus-based templates to guide analysis, examine local practices and institutions, weigh baseline alternatives (such as refining existing fishing rights), or present verifiable measures of costs and benefits to evaluate outcomes within an agreed time frame. The open-endedness of MPA designation and EBFM to achieve undefined public goods provides entry for advocates whose actions impose costs on others with no clear accountability.

Without consensus on what EBFM entails, it is a high-sounding but empty and potentially dangerous concept that allows for broad discretion by advocates. What is the evidence that drives designation of MPAs in a particular location or of a particular size? Are they designated in response to deterioration in fishery and ecosystem quality? Are they pre-emptory and, if so, based on what evidence delivered by what parties? What are the levels of uncertainty associated with the evidence? Are local fishery and ecosystem management practices canvased and considered? If so, is it feasible to work within prevailing institutions and practices if there is credible evidence that ecosystem values are deteriorating? Indeed, where such local arrangements exist, could the MPA and EBFM undermine existing local institutions and practices that offer important value? An overview of two MPA approaches reveals no apparent weighing of any of these questions (see Ministry for the Environment, n.d.; NOAA, n.d.).

Although most fisheries elsewhere in the world lack the property rights that exist in New Zealand, parallel insights are gained from the United States Endangered Species Act 1973 (ESA). This law impinges on the property rights of private landowners by prohibiting habitat destruction where any endangered species has been located. The law did not require alternative approaches to species protection nor did it consider the impact on the property rights of affected landowners as takings. Numerous detailed cases are available (Dolan, 1992; Stroup, 1995; Mann and Plummer, 1995; Seasholes, 1997; Ruhl, 1998; Lueck and Michael, 2003). The law actually weakens prospects for achieving the environmental goal because it shifts the incentives. Landowners take actions that they otherwise would not to degrade habitat so that they avoid losses of asset values. These distorted incentives contribute to the extremely poor performance of the law. Of approximately 1600 species listed, only around 33 have been successfully delisted. The entire process is politicised because advocates bear no costs whereas regulated parties bear all of the costs, and there are no bases for collaborative progress. Property rights allow for such negotiations because there a basis for trade. Conservation easements, for example, allow for voluntary, agreed-to adjustment in practices to achieve environmental goals while maintaining the integrity of property rights.

Within fisheries, there are also insights for New Zealand from case studies of recently expanded MPAs off the north-east coast of the United States where large areas have been placed under no access/no take for commercial fisheries. Analysis would include the arguments and evidence underlying the MPAs, the identity of advocates, bureaucratic agency involvement, and the impact on existing fishing management and fishing communities.

Unregulated Growth of Sports Fisheries

The unregulated growth of sports fisheries presents a direct threat to the property rights held by Māori under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992 as well as the integrity of the QMS in general. The gradual, effective reallocation of Māori fishing rights to new fisheries, such as southern bluefin tuna, is inconsistent with the guarantees of the Deed of Settlement 1992. Moreover, the expansion of the commercial sports fishing sector reduces incomes and impacts on cultural values for Māori, and catch for other QMS quota holders. As sports-fishing fish mortality rises, the basis for quota as a property right is weakened as is the incentive for fishers to adhere to it (Libecap, 2014; Deacon et al., 2013).

Even though the United States fishers do not hold property rights as in New Zealand, many do have catch shares or adhere to other fishery management practices. These may be at risk, along with the livelihoods of commercial fishers, from growth of the sports-fishing sector. The Modern Fish Act, a fisheries management bill, passed by the United States Senate in 2018, grants broader access for recreational and sports fishers in United States fishery management waters. As such, it potentially weakens an already tenuous fishery sector (Bittenbender, 2018).

Project Objectives

1. Analyse the introduction and expansion of MPAs to meet EBFM goals and their impact on Māori property rights agreed on in the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

2. Analyse the effective reallocation of harvest of southern bluefin tuna to sports fishing outside the QMS as it affects the property rights held by Māori under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

Project Activities

Worldwide Expansion of Marine Protected Areas to Achieve Ecosystem-based Fishery Management

1. Review the current debate about EBFM around the world. Taking as a starting point the study by Trochta et al. (2018), we will look for other cases in which the EBFM is being considered. We also will examine the literature on EBFM to identify key terms, trends, data, outcomes, advocates, and local practices addressing ecosystem management.

2. Review empirical evidence regarding MPAs in other countries. Potential documents include those associated with fisheries in waters off Australia, Ecuador, Chile, and the United States. All these countries have more protected marine areas per square kilometre according to the registry of the International Union for Conservation of Nature (IUCN). We will review cases in the United States north-east, the region with the highest percentage of regional waters in some form of MPA (approximately 23 percent of United States MPAs). President Obama declared a fully protected area in the Atlantic Ocean of 4913 square miles off the New England coastline. One of the most affected sectors is the red crab fishery, which has been certified as sustainably managed by the independent Marine Stewardship Council (Eilperin, 2016).

3. Examine the evidence regarding establishment of MPAs worldwide. Are there conclusive indicators of ecosystem deterioration? What is the nature of the data? Who collected it, and how much uncertainty is associated with the evidence? What evidence exists regarding outcomes? Are MPAs successful?

4. Are MPAs worldwide established ex post or pre-emptively? If the latter, what is the basis for such declarations? Who are the advocates? What agencies – international and national – administer the MPAs? Are the agencies active in designating MPAs, and is there a regulatory mandate result? Are local institutions and practices considered in designating MPAs? Are local fishers and communities involved in the process of MPA selection and designation? What was the experience of fishers (e.g., in terms of catch and income) after MPAs were implemented?

5. Make an inventory and review cases of the taking of property rights and indigenous rights in fisheries and other resources. We will start with reports from non-governmental organisations such as Conservation International (Painemilla, 2010) that explored cases around the world of indigenous communities, their rights, and their resource management. Other potential cases include the experience of indigenous communities in countries in Latin America (Chile, Peru, Bolivia, and Brazil).

New Zealand Expansion of Marine Protected Areas to Achieve Ecosystem-based Fishery Management

1. Examine how EBFM and expansion of MPAs interacts with existing law regarding fishery management, ecosystem preservation, and Māori property rights. Legislation includes the Marine Reserves Act 1971, the Resource Management Act 1991, and the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

2. Gather data on existing MPAs and those proposed for expansion to include clear, measurable objectives. What factors underlie designation: ecosystem damage or pre-emptory designation? Who are the advocates? What agencies will manage the MPAs? What are the criteria used for judging effectiveness? Who will determine effectiveness? What is the nature of accountability and review?

3. Evaluate the potential effects on Māori. Addressing this issue requires consideration of stock and fishing location projections, historical fishing patterns, cultural factors, ecosystem protections, and determination of economic and cultural values at stake along with legal requirements for secure property rights under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992. This research may involve interviews with key members of the Māori fishing sector.

Gradual Reallocation of Fishing Rights to Southern Bluefin Tuna from Māori and Others within the QMS to the Commercial Sports Fishing Sector

1. Review the United States Gulf Coast redfish fishery where commercial and sports fishing compete. In the United States, sports fishing for redfish is largely unregulated, and its share of the fishery is growing. Deacon et al. (2013) show how arbitrary share adjustments can lead to an unravelling of the entire quota management scheme.

2. Research the Snapper 7 fishery, a case of a New Zealand recreational fishery where the government is increasing the share of the total allowable catch (TAC) for the recreational fishing sector. This case study has existing information and data that can be analysed for insights regarding the southern bluefin tuna fishery.

Project Implementation

In implementing the proposed research, we considered various options. We chose to focus on the peer-reviewed literature and experiences elsewhere and direct the findings to the specifics of New Zealand. There were several reasons for this approach.

1. If we could show that the problems encountered in New Zealand were reflected broadly in the peer-reviewed literature, then there would be additional credibility for the concerns raised regarding the expansion of MPAs, related EBM within them, and the relatively unregulated expansion of the sports/recreational fishery in New Zealand as they affect Māori property rights granted under the Deed of Settlement 1992 . The alternative option of focusing on the specific case of New Zealand and relating it to other countries' experiences as far as possible within the timing and budget of the grant would not bring in the perspectives and background of this broader, peer-reviewed literature. It would also potentially make the New Zealand case appear idiosyncratic rather than symptomatic of more general problems that should be addressed in New Zealand and elsewhere.

2. This approach utilised the comparative advantage of the University of California, Santa Barbara (UCSB) research team. UCSB is a leader in fishery research and regulation and the design and implementation of MPAs and EBM. Many of the key authors in the peer-reviewed literature are on the faculty there or known to them. Accordingly, it was possible for the UCSB research team to interact with those faculty members to get references to consider, people to contact, and key issues to address. This activity required a major allocation of time and resources that could not be devoted to more specific New Zealand issues. At the same time, it provided a broad perspective on the problems in New Zealand, showing them to be part of major worldwide issues that require revisiting and reform. One member of the UCSB research team spent three weeks in New Zealand this year gathering information and meeting key parties. Additional research using publicly available information was also carried out to document relevant characteristics of MPAs as established under the Marine Reserves Act 1971. It should be noted that this publicly available information may not provide a complete picture of data held by government agencies that would be relevant to this report.

3. Application is pending to access all marine reserve advice papers held by the Department of Conservation and to access marine reserve concurrence advice prepared by Fisheries New Zealand and its predecessor organisations (e.g., the Ministry of Fisheries) to complete this analysis. Summary information and conclusions drawn in the main report should be read subject to further information that may come to light from these internal government files, which may enable a more in-depth analysis of the specifics of the issues. Information publicly available is sufficient, however, to reveal that what is happening in New Zealand is happening elsewhere and there are important lessons to be learned.

4. With the background research provided here, we anticipate that it would be feasible to delve more deeply into the details of the proposed New Zealand expansion of MPAs, EBM, and the sports fishery and their impacts on the property rights held by Māori under the Deed of Settlement 1992 and the strength of the QMS more generally. A more in-depth analysis of the details within New Zealand would, however, require further resourcing.

Outputs

The project has three outputs, compiled as one publication, as follows:

i. EBM/MPA worldwide report – a worldwide report on ecosystem-based management and MPA establishment as applied to fisheries and impacts on rights allocated in the use of fisheries with a particular focus on indigenous rights;

ii. EBM/MPA New Zealand report – a report examining EBM/MPA application in the New Zealand context, drawing from worldwide experiences;

iii. Recreational sports fishing report – a report examining impacts of reallocation in selected United States fisheries and lessons for New Zealand with reference to examples in the southern bluefin tuna and Snapper 7 fisheries.

Key Contributors

Gary Libecap was commissioned to lead this analysis and will be the primary author of all reports. Dr Gary Libecap is Distinguished Professor of Corporate Environmental Management in the Bren School of Environmental Science & Management and Distinguished Professor of Economics at the University of California, Santa Barbara. He is also Research Associate at the National Bureau of Economic Research in Cambridge, MA, Senior Research Fellow at the Hoover Institution, Stanford University, and Senior Fellow at the Property and Environment Research Center, PERC, Bozeman, Montana. He was Pitt Professor of American History and Institutions, Cambridge University, Economics Faculty and Saint Catharine's College, 2010-11. He received his PhD from the University of Pennsylvania and his BA from the University of Montana. His research focuses on the role of property-rights institutions in addressing the open-access losses for natural resources such as fisheries and freshwater, as well as the role of water markets in encouraging efficient use and allocation.

Michael Arbuckle is providing New Zealand-based technical support to Dr Libecap. Mr Arbuckle holds an MSc (Hons) and has over thirty years' experience in fisheries governance, including being the General Manager at the Ministry of Fisheries between 2000 and 2005 where he was responsible for all fisheries management advice and led a significant expansion of the species encompassed within the QMS. He has recently returned from working for ten years as a senior fisheries advisor at the United Nations and World Bank, designing fisheries investments in Africa, the Middle East, Asia, and the Pacific. He is an Independent Director of Seafood Innovations Limited. He is a lead author of The Nature Conservancy's report *Learning from New Zealand's 30 Years of Experience Managing Fisheries Under the Quota Management System.*

Chester Lindley is the main researcher assisting with the project. Mr Lindley has recently completed a Master of Science degree in environmental science and management (coastal marine resource management) at the University of California, Santa Barbara. He has previously engaged in researching United States Census data and property-rights law to see how property rights in the United States were assigned from the colonial period to the close of the frontier and how these rights impacted on human migration and economic development. He is working for the California Fish and Game Commission, Sacramento.

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Output 1:

Marine Protected Areas and Ecosystem-based Management – A Critical Global Overview

Introduction

In the past twenty-five years, there has been growing international interest in the designation or expansion of areas of the world's seas as marine protected areas (MPAs). These are often associated with prescribed fishing practices and other exploitation restrictions such as ecosystem-based management (EBM). Both are advocated as being needed to safeguard global biological/ecological habitats and species thought to be at risk from excessive direct or indirect human utilisation. The Our Ocean conference in Malta, October 2017, outlined MPA target coverage of 10 percent of the world's ocean areas by 2020 with subsequent expansion to 30 percent (European Union, 2017; Wood et al., 2008).

This review of MPAs and EBM is drawn from a comprehensive summary of the literature, where many of the papers examined are summaries themselves or have extensive bibliographies. Many of the papers included were recommended by faculty at the Bren School of Environmental Science and Management at the University of California, Santa Barbara, who have been active in the designation of MPAs. Others were recommended by colleagues at the University of Washington and the University of California, San Diego, who are actively involved in evaluating fisheries management. More broadly, this review targeted literature based on search terms around MPAs, EBM, no-take reserves, ecosystem-based fisheries management, indigenous peoples, and indigenous rights. The literature includes MPA, non-governmental organisations, and international and national agency websites, recent peer-reviewed academic papers from economic and ecological disciplines, and policy-related white papers or grey literature. The conclusions drawn are based on this review. The focus is on institutional arrangements regarding the access and exploitation of marine resources: their origins, including proponents and opponents and the political process underlying them; the distribution of costs and benefits; the incentives they create for resource use, including conservation; and the trade-offs they impose on humans.

Ecosystem-based management is a broadly defined management strategy to encompass aspects of the marine environment that may not be considered in traditional fisheries management practices, such as effort controls in single- or multiple-species fisheries. The notion is that these controls may be too narrow and that broader habitat protections will be missed by fishers without EBM. EBM ostensibly is motivated by the 'best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure, and function' that would not occur under customary fishery regulation (Christensen et al., 1996). EBM focuses on multiple objectives and is claimed to weigh the trade-offs between different management approaches (McLeod and Leslie 2009).

The review below, however, does not reveal trade-off analysis to be common using standard cost-benefit analysis and observational data. Moreover, the broad definition of EBM with little consensus on how to operationalise it leads to confusion about the way policies might achieve the principles of EBM or their impact on fishers (Ounanian et al., 2012; Trochta et al., 2018). The complexity and intensive planning required for EBM, along with the lack of practical tools available to managers and fishers, has led to relatively few actual instances of EBM being achieved through policy intervention (Convention on Biological Diversity, 2007). This is not to say that fishers and fishing communities do not already carry out the principles of EBM within existing management regimes, which is an issue addressed below.

Because MPAs, by their very nature, are executed as a command-and-control government regulation that defines ocean boundaries and outlines permitted uses within them, they are often promoted as a tool to achieve some of the loosely defined goals of EBM regimes. MPAs establish the spatial dimensions within which EBM management can be implemented by prohibiting or constraining specific resource-use activities thought, generally by fishery biologists and ecologists, to damage the ecosystem. Without these constraints, standard fishing practices, especially under open-access conditions, may inflict externalities, such as excessive bycatch of non-target species or harm to seabirds, seagoing mammals, corals, and the seabed. As noted below, however, the literature does not address the baselines for imposing EBM regulation, timelines for recovery of the ecosystem, trade-offs imposed, or why existing fishing and other resource-use practices and institutions cannot be adapted to achieve ecosystem goals.

No-take marine reserves, a particularly restrictive type of MPA, are claimed to be associated with EBM goals of increased density, biomass, and diversity across different trophic levels (Halpern, 2003). However, the extent to which these no-take reserves, and MPAs more generally, can address the broad goals of EBM is uncertain as MPAs typically only encompass a small fraction of any given ecosystem. Accordingly, it is unlikely that they will adequately address the broad impacts that EBM seeks to address on a much larger scale, and there is little empirical evidence that analyses the importance of MPAs in achieving EBM goals (Halpern et al., 2010). Moreover, EBM goals are not stated in a clear and measurable manner so as to allow for validation or determination of other regulations or MPA adjustments, should the objectives not be met. There is little discussion in the literature on long-term uncertainties in achieving biological goals. This ambiguity in design and goals creates uncertainty for resource users who are or might be critically affected by EBM. Ecosystem conditions necessarily vary case by case, so that broad, one-size-fits-all EBM prescriptions may not be appropriate for both differing ecosystems and fishing practices that occur within in them (Halpern et al., 2010). Finally, except for a few cases, EBM regimes do not take advantage of the latest incentive-based management fishery tools that create vastly different incentives for marine protection than do standard limited entry and effort controls. Established in a collaborative manner with fishers who are part of incentive-based or rights-based systems described below, EBM can be much more realistic, effective, and welfare-enhancing with fewer distributional conflicts.

The overwhelming emphasis on MPAs and their application in EBM arises from government officials (international and national), members of non-government organisations, and related lobbyists (academic and non-academic consultants). Their efforts are aimed at modifying existing marine resource access and use, often as prescribed through multinational treaties drafted at international conference sites and signed by national government officials. They typically are not spearheaded by those most directly dependent on the marine resource: local fishers and residents of adjacent communities. The rare instances where local communities are involved in the planning and management process, cases such as Apo Island Reef Park in the Philippines and the SGaan Kinghlas-Bowie Seamount (SK-B) in Canada, are often referenced as major success stories of MPA implementation (Russ and Alcala 1999; Gaines et al., 2010; Haida Nation, 2018). Indeed, the EBM and MPA literatures do not address the importance of incorporating the opinions and practices of incumbent fishers and their communities; the trade-offs fishers and their communities face currently and under proposed regulations; how the interventions might affect standard or customary fishing practices; how the practices blend or do not blend within existing law regarding fisheries and related resources; and how the costs and benefits asserted to accrue by advocates would materialise and when (Halpern et al., 2013).

Indeed, because both MPAs and EBM involve new central-government regulation, they run counter to ongoing fishery management trends. Although, most fishery regulation since World War Two has involved command-and-control restrictions on entry and inputs (seasons, vessel size, equipment), in light of their often-observed and documented failure to revive fishery economics and stocks, rights-based or incentive-based systems have been adopted. These involve enlisting actual fishers directly in management through the assignment of individual or group property rights. Fishers have the most information about and the greatest incentives to care for the marine resources on which their livelihoods and communities depend. Rights-based systems include setting annual total allowable catch amounts and the distribution of shares or quota as a property right to individuals or companies. Additionally, there are group rights to specific areas or territorial use rights for fisheries (TURFs). Where these rights are secure, long-term, and tradable, as described below, they have brought important gains over previous regulation.

The provision of global public goods through ecosystem protection is emphasised by advocates of MPAs and EBM. There is no specificity as to what global public goods are provided by action at the country level, whether the MPA/EBM is large enough to impact global variables, or whether or not such action is welfare-enhancing for its citizens. Such outcomes are assumed to be the case. Nevertheless, adoption, management, and durability of MPA/EBM depend on how they affect country constituents and on the actions of internal political coalitions.

Because most MPA/EBM proposals are vague and do not include measurable outcomes, timelines, or policy adjustments, they potentially provide for maximum regulatory agency discretion. Because they are so broad and vague, the proposals do not facilitate constituent assessments of how policies might play out, what their private costs might be, and what ensuing benefits might accrue. Affected parties must form expectations based on limited information and high levels of uncertainty. These conditions, in turn, reduce any realistic constituent benefit/cost calculation. Although the proposals are driven by global objectives formed by advocates, their implementation relies on actual national budget allocations and political durability, which depend on constituent support. If it is not in the interest of those with the most stake in the ocean resources to back MPAs and EBM, they will not do so.

Review Findings

MPA Definition

There is no uniform definition of MPAs. Marine protected areas involve a variety of different management interventions, sizes, locations, existing ecosystem conditions, and include no-take or limited-use areas. Definitions are presented with a broad brush. One definition states that an MPA is a 'clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values' (Dudley, 2008 quoted in Spalding and Hale, 2016; Bander et al., 2015). Another definition includes natural marine areas, with permitting and non- permitting use, tidal and subtidal coastal areas, to achieve goals of biodiversity conservation, economic resources and species protection (UNEP-WCMC and IUCN, 2016). The vagueness and variation in MPA definition and goals make critical analysis of the objectives of MPAs and impacts on national constituent groups, where actual policy must take place, extremely difficult. It also hinders the design of alternative approaches, if warranted, to achieve reasonable biological goals. There are few thresholds or templates for reassessment.

EBM Definition

As with MPAs, EBM is broadly defined to include management strategies for sustaining ecosystems (Christensen et al., 1996; Pikitch et al., 2004). It often is associated with MPAs because these are spatial areas where regulatory controls can be implemented and monitored. For example, the National Marine Sanctuaries in the United States, that cover over 375,000 square kilometres of ocean, are designed to protect ecosystems within them (Lindholm and Pravia, 2010). Commercial and recreational fishing within them must occur within the framework prescribed and monitored by the National Marine Fishery Service. Even so, for EBM there is no general agreement on management strategies, types of ecosystems that might benefit, timelines, or performance criteria (Trochta et al., 2018). As with MPAs, this vagueness leads to a variety of interpretations and objectives in prescribing restrictions on resource use that will affect users and their communities, who bear far more precise costs. The ambiguity of definition invites extensive regulatory discretion and associated uncertainty for users.

Durability of MPAs

Although MPAs and EBM are called for by members of broad international organisations, such as United Nations agencies, and worldwide non-governmental organisations to provide global public goods, they must be designed and implemented at the country level, affecting country citizens, budgets, and use of natural resources. The lack of precision in MPA and EBM definitions in local policy discussions leads to distrust and conflict, which have potential economic and political costs. Advocates have general policy objectives that they promote, but they typically do not bear direct private costs of the restrictions imposed by MPAs and EBM. By contrast, those who will bear the private costs of implementation of MPAs and EBM, with unclear benefits, will have very different objectives and incentives in mind. This does not bode well for long-term political durability of MPAs or of the economic and social returns from national marine resources affected. Ongoing disagreements over MPAs and their potential distributional effects in United States state waters, as well as north-eastern United States federal waters, addressed below, illustrate the costs of discord between fishers and members of regulatory agencies.

Criteria for Establishment of MPAs

Similarly, there are no generally understood criteria for the establishment of MPAs. Some may be pre-emptive, inserted into areas of partially pristine ecosystem conditions, such as the Great Barrier Reef Marine Park (Day, 2002; Great Barrier Reef Marine Park Authority, 2014). Others are opportunistically created by advocates in areas of little current human exploitation or constituent involvement or reaction, such as the Marianas Trench Marine National Monument (NOAA, n.d.b). Because of limited information about such areas and the desire to lock them from human exploitation, the economic returns and potential public goods associated with the use of valuable resources within them will be denied to current and future generations. Although MPAs typically are established to protect ecosystems for future generations, the lack of specificity in criteria definition makes it difficult to evaluate the effects of any MPA on later human populations (Pendleton et al., 2018). Positive predicted outcomes are not based on rigorous trade-off analysis. Additional MPAs, such as Apo and Sumilon Islands in the Philippines, may be more remedial, having been established as responses to potential or existing degradation, such as dwindling fish stocks and overall decline in catch per unit effort (Aliño et al., 2002).

Objectives for MPAs

In general, there are few measurable targets for assessing success in achieving MPA goals. The absence of specific, identifiable goals, timelines, and sources of uncertainty or risk, not only make it hard to judge performance, but also add policy uncertainty if the MPA appears to fail. What actions might be taken for management options should the initiative require modification? How flexible are MPA/EBM policies and how do adjustments impact existing and potential users? There are many sources of uncertainty in biological outcomes and not all are controllable within an MPA, particularly for global factors such as rising water temperatures or increased salinity. In other cases, if the MPA appears to successfully meet biological objectives, can constraints on use be relaxed or adjusted? If so, when and how? Detail on such potential outcomes is required for clearer assessment of the trade-offs born nationally with MPA establishment.

MPA Links to EBM

MPAs that are very remote and in areas of no major existing human activities (fishing, tourism, mining) may involve fewer cost-benefit conflicts. Such remote MPAs appear to be larger and are classified as large-scale marine protected areas (LSMPA), being 150,000 square kilometres or more. There are sixteen LSMPAs that make up Big Ocean, a collective support network to design and manage LSMPA (Lewis et al., 2017). Inshore or close-offshore MPAs, however, are more likely to conflict with existing human use, particularly since worldwide MPAs cover 10.2 percent of coastal and marine areas under national jurisdiction (UNEP-WCMC and IUCN, 2016). Some MPAs prohibit all fishing or other human extractive activities with no-take zones. Globally, 1.6 percent of MPAs are fully protected from any form of use (Lubchenco and Grorud-Colvert, 2015), whereas remaining areas involve some degree of multiple use, particularly by fishers and/or tourism. These MPAs generally regulate human activities through centralised EBM.

Cost-Benefit Analyses

Cost-benefit analyses of MPAs and related EBM to assess trade-offs, particularly as they include precluded or restricted human activities, such as fishing or mining, are very rare in the literature. Where cost-benefit analysis occurs, it typically uses model-based simulations and not observational data. For example, in southern California, a bio-economic model was used to assess how regulated fishing pressure might increase profits with a strategically designed MPA network (Rassweiler et al., 2012). On the other hand, Fiji's locally managed marine area network appears to have provided economic gains to villages through improved fish catches and tourism employment (Brander et al., 2015). This is an unusual example, both in terms of the involvement of locals in design and implementation and in outcomes. Trade-off analysis could be far more prevalent because numerous MPAs have been in operation for sufficient time to provide data and to allow for cost-benefit studies.

To effectively assess trade-offs facing citizens when countries consider implementation of MPAs, the following parameters must be included in cost-benefit analysis: relevant interest rates for discounting future costs and benefits; expected price changes for the affected natural resources; risk and uncertainty both for biological outcomes and for economic impacts; and identification of the country citizens most likely to be affected. This is not the same as cost- effectiveness analysis that sometimes is presented to justify MPAs (Halpern et al., 2013). Such analysis is aimed at minimising the costs of proposals that have already been decided. In contrast, cost-benefit analysis is aimed at determining whether or not a policy should be adopted, relative to other national economic, political, and social objectives.

Integration with National Laws and Indigenous Rights

MPA proposals are generally presented in isolation from national policies and legal obligations. Nevertheless, they involve costs and potential benefits and, therefore, must be weighed in light of other national objectives and responsibilities. For example, fishing communities, and especially those with indigenous populations, often perform poorly relative to the national socio-economic criteria. Indigenous populations also have treaty guarantees that may be compromised. If MPAs inflict added costs, then these outcomes would be inconsistent with other policies.

Indeed, MPA and EBM discussions rarely acknowledge the incentives of indigenous and other local peoples who depend on fisheries for their livelihoods and may incorporate the surrounding ecological and terrestrial systems into norms and customs that protect the ecosystem in a manner that is quite different from open access (Guénette et al., 2000). In a literature review by Ban and Frid (2018), less than 0.5 percent of MPA papers include involvement of indigenous peoples in design. Nevertheless, the practices of indigenous and other local parties can be an alternative to MPAs, achieving more ecosystem goals at lower cost. They are locally based and understood, whereas MPAs and EBM typically are top-down initiatives.

Compensation

Compensation to resource users affected by MPA no-take or highly restricted access regulations is extremely rare. Even if compensation or alternative economic opportunities were proposed by MPA advocates, they may not materialise. This was the case with the Galapagos Marine Reserve where fishers were promised compensation in the form of licences and help in transitioning to new tourism industries that would grow from the creation of large no-take zones. However, this compensation promise was diluted by a dramatic increase in the number of fishers after no-take zone implementation, and fishers felt 'cheated' as compensation took longer than expected, which led to an overarching distrust between fishers and regulators advocating for no-take reserves (Castrejón and Charles, 2013). If parties are made worse off, they may resist MPA/EBM management and these conditions raise the costs of implementation, management, and enforcement of MPAs. As discussed below, such higher costs question the political durability of MPAs, which is especially important given that they are designated to provide long-term future benefits (Weigel et al., 2014).

The Great Barrier Reef MPA rezoning in Australia that took place in 2004 is an unusual case of direct compensation to fishers. During the effort to rezone the Great Barrier Reef and to dramatically increase the area deemed no-take, the Australian government agreed to compensate commercial fishermen who were adversely affected by the new zoning (Olsson et al., 2008). The cost to fishermen from the closures was originally estimated at around AUS$14 million a year (McCook et al., 2010). Assistance included the costs of transitioning out of the industry, compensation to those who lost jobs, paying for financial/legal advice, and helping the communities impacted by the loss of fishing (Macintosh et al., 2010). Between July 2004 and October 2008, 1783 applications for assistance were approved, with a total cost of AUS$205 million (Macintosh et al., 2010). While this financial commitment to assisting commercial fishermen is considerable, it is estimated in total that the income from tourism to the Great Barrier Reef is around thirty-six times greater than commercial fishing, and that ratio was increasing during this time (McCook et al., 2010). It is important to underscore, as this study shows, that the tourism benefits did not necessarily accrue to fishers or their communities. Distributional effects must be addressed in any cost-benefit analysis.

Absence of Trade-off Analysis

The absence of economic trade-off analysis, such as cost-benefit analysis, is often justified through arguments that ecosystem values are extremely difficult to assess without extensive data (Garces et al., 2013; Rosales, 2018) and that, in principle, they should not be assessed in economic terms because they involve non-human values. In contrast to these claims, there are long-standing, established methods for valuing non-traded resources in economics, including contingent valuation, travel-cost analysis, hedonic studies, and benefit transfer. The failure to conduct such studies, implicitly assigns an infinite value to the biological goal as a global public good, which is unlikely to be the case at the local or national level. Defining, maintaining, and enforcing MPAs and EBM is costly and requires real resources. Although international non-governmental organisations and intergovernmental organisations may provide start-up funding, most governments must rely on country allocations over the long term, particularly given the ambitious objectives of expanding MPAs to cover up to 30 percent of the ocean surface. Budget allocations occur through the political process involving politicians, agency officials, and lobby groups. There are many competitors for funds. High-cost, unpopular or controversial activities consume political capital because trade-offs have to be negotiated and incorporated into policy. MPAs and EBM that impose disproportionate costs on key constituents are unlikely to be durable, undermining the biological goal.

The MPA literature generally does not incorporate the findings of the collective-action literature in the social sciences, which reveals that policies are long lasting when costs and benefits are distributed proportionately among parties. If not the case, those who bear more costs than benefits are made worse off and will resist, and those who bear more benefits than costs will promote. This setting results in conflict that raises the costs of any environmental policy. MPAs are presented as providing global public goods, so the notion of disproportionate cost is not addressed because all parties are presumed to benefit. Although there may be generalised public-goods benefits, more narrow policy benefits and costs are far more apparent and particular to specific national constituent groups. Differential distributional outcomes critically mould support for public-goods provision and for policy resilience.

The issue of disproportionate potential costs/benefits distributions in MPAs/EBM arise because advocates generally are members of international non-governmental organisations or intergovernmental agencies, along with national politicians and regulatory agency officials. Rarely are these initiatives inaugurated by existing users. Given the gains that are often described with MPAs, the question arises as to why local users resist. They do so because the benefits are uncertain, generalised, and long term, whereas imposed costs are far more immediate and clear. Advocates seek to implement their values, generally framed as global public goods, but do not bear direct costs. In contrast, parties whose fishing or other resource-use practices are curtailed bear direct costs. At the country level, interest-group politics moulds outcomes. Fishers can be at a disadvantage as they often are poorer, with less education, and are spread across multiple locations and fisheries, which are factors that raise the costs of mobilising lobby groups. By contrast, members of non-governmental groups, consultants, and elected and unelected government officials have higher incomes and higher levels of education and are adept in the policy arena.

Baseline Assumptions

The baseline alternative for MPAs and EBM is not defined. When open access and the race to fish dominates, then short time horizons prevail with excess labour and capital devoted to the fishery, low profitability, depleted target stocks, high levels of bycatch, and little ecosystem preservation. MPA/EBM discussions typically point to these conditions as the source of human degradation of biological systems and justification for MPAs. But open access or traditional regulatory practices are being replaced by local, rights-based systems that result in different incentives for resource use. As noted above, MPAs and EBM generally move in the opposite direction. Alternatives to achieve agreed ecological goals using rights-based systems can be timely, less contentious, and more effective.

Discussion of Key Issues

Movement towards and Advocates of Global MPAs and EBM

Advocates assert that MPAs and EBM are essential for current and future generations and the overall health of the earth's ecosystem. These initiatives aim to provide broad global public goods, and the lobby efforts behind them are coordinated by major international organisations. Representative of this is the call for action at the 2017 United Nations Ocean Conference:

> *We, the Heads of State and Government and high-level representatives, meeting in New York from 5 to 9 June 2017 at the United Nations Conference to Support the Implementation of Sustainable Development Goal 14 of the 2030 Agenda, with the full participation of civil society, and other relevant stakeholders, affirm our strong commitment to conserve and sustainably use our oceans, seas and marine resources for sustainable development. We are mobilised by a strong conviction that our ocean is critical to our shared future and common humanity in all its diversity. As leaders and representatives of our Governments, we are determined to act decisively and urgently, convinced that our collective action will make a meaningful difference to our people, to our planet, and to our prosperity* (United Nations, 2017).

The related call for ecosystem-based management or an ecosystem approach to fisheries (EAF) is provided by the Convention on Biological Diversity (CBD). As early as the Second Conference of Parties in 1995 CBD members affirmed, 'the ecosystem approach should be the primary framework of action to be taken under the Convention' when addressing the sustainable use of biological resources and addressing biodiversity and socioeconomic and cultural factors (Convention on Biological Diversity, 1995). More specifically for marine EBM, CBD advocated at the Strategic Plan for Biodiversity, Conference of Parties 10, 18–19 October 2010: 'By 2020 all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally and applying ecosystem based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems and the impacts of fisheries on stocks, species and ecosystems are within safe ecological limits' (UNEP, 2010; NOAA, n.d.a.).

International efforts that underlie country MPA designations, generally within country exclusive economic zones (EEZs), include provisions of the United Nations Framework Convention on Climate Change, whereby member nations committed to designate new marine protected areas by 2009; the 1992 UN Conference on Environment and Development (UNCED) and the Convention on Biological Diversity (CBD) that encouraged the use of protected areas or area-based closures; and the 2002 World Summit on Sustainable Development (WSSD), where MPAs were placed at the top of the agenda for implementation in order to achieve sustainability. Among non-governmental organisations, prominent advocates for MPA designation and expansion are the Pew Charitable Trusts and Oceans Initiatives and Conservation International (Fathom, 2016). Other supportive non-governmental organisations include Birdlife International; Blue Marine Foundation; CORDIO East Africa; Global Ocean Trust; International Union for Conservation of Nature: IUCN; Commission on Protected Areas, Marine Conservation Institute; Marine Affairs Research and Education: MARE; Oceana; Ocean Unite; Oceano Azul Foundation; Rare; The High Seas Alliance; The Nature Conservancy: TNC; Waitt Foundation and Waitt Institute; Wildlife Conservation Society: WCS; and World Wide Fund for Nature: WWF.

For the most part, these non-governmental organisations are well funded and organised, and the various United Nations and other international conferences provide opportunities for advocates to meet to identify areas they believe are of critical need, to present action proposals, and to launch country-level efforts for implementation. For example, the United Nations Convention on the Law of the Sea, Subcommittee on Oceans and Coastal Areas and the 2002 Word Summit on Sustainable Development led to the creation of UN-Oceans in 2003 to coordinate international actions towards ecosystem sustainability (United Nations Sustainable Development Goals, n.d.).

As noted above, MPAs can involve vast areas of sea being set aside, effectively placing them off limits to entry by fishers and other resource users. Other MPAs allow for multiple uses, including fisheries, recreation, and tourism. Where fishing is allowed, it is often to be regulated in a variety of ways to achieve ecosystem goals through EBM. Some 14,688 marine protected areas have been designated in the world's oceans and coastal marine areas under national jurisdiction (UNEP-WCMC and IUCN, 2016). MPAs vary widely in size, location, and nature. They range from many small MPAs <1 square kilometre throughout the Philippines to the 1,500,000 square kilometre marine monument north-west of Hawaii (Alcala, 1988; Obama, 2016). There are many MPA summaries and databases. One summary is by Wells et al. (2008), and another is the Atlas of Marine Protection (http://www.mpatlas.org/) prepared by the Marine Conservation Institute in Seattle that claims to be the most comprehensive.

Absence of Verifiable Objectives and National Application: Interest-group Politics and Agency Incentives

As described above, neither MPAs nor EBM are defined in uniform, clear ways as to what the concepts mean, when and where they should be implemented, and what they are to deliver (Trochta et al., 2018). The biological objectives for MPAs and EBM are not presented in measurable, verifiable ways. These would include timelines, complexity and interaction within relevant ecosystems, exogenous (uncontrolled) factors, and uncertainty. Such factors allow for determination of goal attainment, the actions that must be taken, and the costs that must be incurred. Both concepts are couched in broad terms for public-goods provision with little specificity for implementation, evaluation, and assessment. The details for achieving them are left to country politicians, agency officials, consultants, and national members of international non-governmental organisations.

After international proclamations are made, member delegates return to their home countries to outline what they believe to be the underlying obligations for implementation. National application requires adapting generalised global public-goods objectives to specific cases, and as this occurs, the potential effects on country constituent groups become far more precise. Latitude for implementation exists with local politicians and administrative agency officials. These parties decide how to respond with input from interested lobby groups, including country representatives at the international forums, members of supportive transnational non-governmental organisations, and consultants. These are also the people with the most direct knowledge of the global effort. In political and administrative deliberations, other constituent groups also mobilise, such as those from industry and community groups that may be affected. Because these groups typically were not part of the initial international MPA/EBM effort, they assemble later in the policy development process, which potentially places them at a disadvantage compared with advocates.

Constituent response depends on the way policy specifics unfold. Only then can those likely to be directly affected assess how their welfare may be impacted. Given that national policies are cloaked in globally-beneficial terms, those who perceive direct costs can be portrayed as responding to narrow private interests. Advocates – supportive politicians, agency officials, non-governmental organisation members, and consultants – who do not bear direct costs (that is, their livelihoods generally are not at stake) and have their values advanced can be portrayed as furthering broad social objectives.

Absence of Cost-Benefit Analysis

The global objectives of MPAs/EBM do not call for critical evaluation of trade-offs nor of multiple ways of advancing towards a goal. Further, they do not recognise that in some specific MPA/EBM settings broad ecosystem goals may not be achievable or advance human welfare. As noted earlier, the broad sustainability objectives underlying MPAs are not defined with sufficient specificity to invite trade-off evaluation.

Actual implementation, however, takes place at the country level where real resources will have to be committed over the long term, due to the nature of the biological goals outlined. This means that durable, supportive political coalitions have to be formed to support country- level policies. Significant economic and social costs, especially to organised constituencies, undermine such political durability. The practical importance of trade-off analysis and policy flexibility, including major modification or abandonment at the country level, is not recognised in MPA documents, including websites from non-governmental organisations, international and national agencies, the academic literature or policy white papers reviewed here.

Country budgets and resources are constrained. MPAs and EBM require definition of areas; species and ecosystems to be included; parties and activities to be excluded or limited; enforcement; and ongoing management. The literature review by Halpern et al. (2010) that links MPAs and EBM in achieving ecosystem protection emphasises that MPAs must be well defined and enforced, especially where fishing pressure is the greatest. These are the same areas, however, where enforcement costs are likely to be highest, so agreement on objectives and compliance with the MPA and related EBM will be critical otherwise resistance will be widespread, compromising the biological objective – an issue not addressed generally in the literature.

National resource requirements depend not only on the complexity of the ecosystem problem, but on constituent support and buy-in to the process. This process of achieving consensus and compromise can take a very long time depending on the magnitude of MPAs/EBM proposed, the degree to which they conflict with established resource-use practices, existing legal institutions, and information on the costs as they unfold. Those parties who will bear costs due to proposed access and use constraints must perceive that these restrictions are reasonable, compensable, or aligned with benefits that are alleged to accrue to them. These parties include fishers and fishing communities or, potentially, mining and other extractive users and their communities whose activities are to be prohibited or constrained. The literature on support for collective action is uniform in the empirical finding that costs and benefits must be proportionately distributed (Ostrom, 1990; Cox et al., 2010; Libecap, 2014). If that is not the case, then those who bear disproportionate costs have incentive to defect or, in the case of MPAs or EBM, to violate policies, raising enforcement costs and undermining ecological goal achievement.

Additionally, local fishers and community members are often the parties with the most information about the ecosystem and how it might respond to different policy recommendations. If those parties are disaffected, then they will be less likely to share that information, hindering policy objectives. In contrast, those who gain disproportionate benefits have incentives to seek more action. These conflicting groups make policies controversial, even those framed as public-goods provision, and raise political conflict within countries. This, in turn raises costs to politicians. Achieving consensus on MPAs and EBM can reasonably involve compensation to parties harmed on net or substantial alteration of policy proposals. Advocates driven by global public-goods objectives can be insensitive to the details of the costs imposed. Even if there are alternative benefits from MPAs, such as greater tourism or recreational opportunities, these benefits may accrue to other constituencies and communities. There will be within-country distributional outcomes, and how these are addressed will affect collective support for adherence to international agreements and their implementation within countries.

In general, rigorous social science approaches are not included in MPA analyses, where biological considerations, not social ones, dominate. Because MPAs are placed within unquestioned global public-goods provision, there are few actual assessments of impacts on fishers and other resource users, even for MPAs that have been in existence for sufficient time to allow for such analysis. Beneficial projections are made for both the ecosystem and humans. These, however, are model-based simulations that are not based on actual observations, and they do not include critical economic criteria. For example, consider Halpern et al. (2013) who argue that in three case studies in central California, Indonesia, and Southeast Asia, social equity, economic return, and conservation are all feasible. They do not perform cost-benefit analysis, but rather cost-effectiveness whereby the predetermined conservation goal is to be achieved at least cost. Cost-benefit analysis in contrast would determine if or how the initiative would be implemented or adjusted. The trade-offs outlined in the paper are between equity sharing of simulated benefits and conservation objectives.

It can be argued that ecosystems do not lend themselves to trade-off evaluation. Their values are broader and non-monetised. This is a legitimate criticism, but the values of ecosystem services have long been assessed through a variety of techniques in economics, ranging from contingent valuation, travel-time cost, hedonic studies, and benefit transfer (Dixon, 2012). As a clever alternative, Kotchen and Burger (2007) measure the costs of a major conservation set-aside involving the Arctic National Wildlife Reserve in Alaska. Costs typically are more directly measured, and Kotchen and Burger sought to calculate a cost value as a benchmark that any ecosystem valuation would have to meet for a benefit-cost ratio equal to one. Their analysis provides an unexpectedly large present value opportunity cost estimate of US$1,141 per adult citizen of the United States for not producing from this terrestrial set-aside. Because no exploitation was underway, there were no adjustment costs that would also have to be addressed.

Achieving a stream of expected costs and benefits from an MPA that meets standard acceptance criteria requires definition of time frames for achieving the biological objective; time frames for constraining human activities; identification of alternative options; determination of relevant discount rates; assessment of the opportunity and transaction costs inflicted on particular parties as well as the benefits achieved by others; and determination of the programmatic costs over time (that will not be independent of constituent-group reactions). These cost-benefit criteria are virtually absent from the MPA/EBM assessment literature. If they were included, then not all MPAs/EBM could be achievable, calling for reassessment, redesign, or abandonment. This is very useful information because it avoids political and social conflict over unattainable policy goals and focuses attention on those initiatives that provide for ecological gains and improved human welfare.

As with Halpern et al. (2013), most analysis of the MPA impacts on user groups relies on simulations of ecosystem responses and the related predicted effects on fishing harvests and profitability. They typically are optimistic in predicted impacts on users. Dalton's (2010) simulations suggest that MPAs increase profitability, especially through strategic placement of smaller MPAs to promote recruitment and growth of fish stocks, for fishers outside MPA boundaries. Another study by Adams et al. (2010), however, recognises that opportunity costs that are unevenly spread can impact users, their support for nature preserves, and the potential for success.

Gallacher et al. (2016) perform an MPA literature review and then apply the general findings to a case: the Lyme Bay, England, marine protected area. This is an unusual example, where actual observational data are gathered for determining economic impacts, as opposed to using simulations. The authors point out that most discussion in the literature focuses on biophysical criteria for establishing MPAs and not on economic or social factors. In the Lyme Bay fishery, the authors found that mobile and mixed-gear fishers, such as scallop fishers, bore the brunt of policy costs and were forced to leave the fishery as their income declined. Recreational fishers, on the other hand, who had access to adjacent areas were not harmed.

In an unusual and useful cost-benefit analysis in the peer-reviewed literature, Pascal et al. (2018) examine the ecosystem benefits of five small community MPAs in Vanuatu in the Pacific and one large government MPA at Saint Martin in the West Indies. The focus is on tourism benefits resulting from improvements in the natural environment related to fish biomass, scenic beauty, protection against coastal erosion, bequest and existence values, social capital, and greenhouse gas sequestration. These benefits are difficult to calculate, and the authors use simulations, surveys, and benefit-transfer methods. They also analyse surveys and catch-per-unit- of-effort data for small-scale fisheries at both locations. The number of observations is small. The benefits are compared to the costs of administering MPAs. Opportunity costs to fishers from MPA restrictions are examined, but their size is not clear. The authors could not detect changes in fish size or variability of harvest. A twenty-five-year time span and 10 percent discount rate are used in the analysis. They also determine that the value of benefits varies between tourism and small-scale fishing. In the end, the authors find that benefits exceed costs, giving a benefits/costs ratio greater than one, and that corresponding rates of return in MPAs at both sites justify the investments in them. The authors caution that these results do not suggest that increasing MPA sizes would result in even greater benefits.

Another uncommon study of the direct effects on fishers from MPAs that initially were predicted to improve profitability is by Guenther (2010). She uses observation data and examines the spiny lobster fishery in the Santa Barbara Channel. Marine managers created MPAs as part of the Channel Islands State Marine Reserve (CISMR) network that spatially limited or prohibited fishing of lobster. One objective of the CISMR was to enhance predator abundance (spiny lobster) to control the rapid increase in prey species (sea urchins) and prevent trophic cascades that threatened kelp beds in the ecosystem as a whole (Ugoretz, 2002). Despite supportive simulations as part of the set-up of the reserves, using panel datasets of five years before and five years after the CISMR designation, Guenther found that kelp cover and sea urchin abundance were far more affected by reef characteristics and tide patterns than by fishing effort. She also estimated a 28 percent loss in individual daily catch associated with denial of access to fishing grounds. Two-thirds of the loss was due to forcing fishers to search for and learn about new areas for fishing.

These possible negative effects are best anticipated by fishers whose livelihoods are directly affected by MPA and EBM policies placed on them. This probably explains why efforts to expand MPAs in central California to include 10–20 percent of coastline up to five kilometres from shore have been very contentious, with conflict between marine conservationists and fishermen (Dalton, 2010). The establishment and maintenance of MPAs requires more than simulations, and it needs gradual measurement of results and flexibility in MPA design, including rejection. Trade-off analysis for MPAs is described in Lester et al. (2013).

Unclear Baselines for Comparison of MPA/EBM Proposals

As described earlier, the MPA literature rarely identifies the baseline setting of concern. Is the proposed MPA in an area of open-access fisheries where timelines are short and there is a race to fish? Alternatively, is it an area with government limited-entry fishery controls on inputs and seasons whereby incentives for ecosystem protection, including bycatch controls, are unclear? Finally, is it within areas with annual allowable catches and shares or quota within them or TURFs? Are all of these areas within the waters of countries that adhere to the rule of law, have enforcement capabilities both for internal compliance and against illegal entry from outside parties, and have mechanisms to elicit the concerns of various internal interests and respond to them?

The counterfactual baseline for MPAs is critical because it determines what options might be considered for achieving the ecological goal and what is feasible. If the alternative is open access, the question arises as to why open access exists. Open access has long been understood to bring important losses (Gordon, 1954; Scott, 1955; Hardin, 1968). An alternative solution for both improvements in fishery outcomes and ecosystem protection could be the assignment of property rights through catch shares (Christy, 1973). Traditional government limited-entry controls and restrictions on inputs fail to fully align incentives and to reduce the losses of the race to fish (Grafton et al., 2000). Moreover, there is no basis for bargaining among fishers in an open-access setting. They cannot easily contract among themselves to halt damaging fishing practices. Except for unusual cases (Acheson et al., 2015), there are no owners or enforcement frameworks for such private contracts. Open access and weak government regulation are likely to coincide with ecosystem damage. A remedy is to reform fishery regulation towards incentive-based or rights-based systems and then incorporate ecosystem concerns in the design. This remedy could be far less controversial and provide broader human and ecosystem benefits than MPAs/EBM.

Indeed, property rights have been assigned through the definition of total allowable annual catch (TAC) and the assignment of shares within them. Where these catch shares or quotas are durable and tradable, they fundamentally change incentives in human exploitation, which is a critical condition not recognised in the MPA literature (Essington et al., 2012). Rights-based systems provide both long-term incentives in harvest and ecosystem protection because the stock depends on the ecosystem. Assessments of catch-share systems generally are positive in terms of improvements in profitability, reduced variance in harvest, and, in some studies, ecosystem protection (Grafton et al., 2000; Costello et al., 2008; Bonzon et al., 2010; Essington et al., 2012; Afflerbach et al., 2014; Thunberg et al., 2015; Birkenbach et al., 2017). Criticism of rights-based systems is generally driven by distributional concerns (Bromley, 2009).

Rights-based systems not only change incentives, but they allow for bargaining within groups. They identify who the quota or catch shareholders are and provide the basis for negotiated arrangements to change fishing behaviour. Such negotiations are not possible in open access or within traditional effort regulation. There can be unlimited numbers of parties and none has legal standing to engage in group negotiations to protect the ecosystem. Catch-share systems, by contrast, are well placed to take on a number of responsibilities to achieve ecosystem goals, including bycatch controls and restrictions on spatial access.

Holland (2018) outlines how rights-based systems were used to achieve biological objectives in the Bering Sea and Aleutian pollock fisheries and Pacific whiting fisheries in Alaska, all under quota systems. These industry-led cooperative arrangements reduce bycatch of non-target species. Caps are set on bycatch, quotas assigned are tradable, and unused bycatch credit can be carried forward. There also are risk pools of bycatch quota among members to reduce the hazard of target fishery closure. None of these actions would be feasible in the absence of a catch-share system of some type. As another example, the British Columbia groundfish trawl fishery, which has been managed with individual transferable quotas since 1997, reduces incidental harm to deep-water corals and sponges through some area closures and transferable bycatch quota. Notably, the actual catch of corals and sponges has remained far below the total quotas set since the programme was implemented and also well below average levels prior to the agreement.

Another property-right regime is spatial, assigning use rights to a group as TURFs or territorial property rights (Christy, 1982; Cancino et al., 2007). Afflerbach et al. (2014) compiled twenty-seven TURF reserves worldwide, suggesting that strong customary tenure systems result in distinct qualities of governance, management, and enforcement, as opposed to government-mandated TURFs. TURFs make up about 10 percent of worldwide catch-share systems (Holland, 2018). Holland argues that such spatial collective rights arrangements can address both open access and ecosystem values. They can be effective where there are many fishers and assigning individual property rights is costly or less effective than having group rights and harvest practices. Costello and Kaffine (2017) provide a model whereby TURFs lower the costs of defining, managing, and enforcing marine conservation. The group internalises the benefits and self-enforces private spatial-biodiversity controls.

Because catch-share and TURF systems can be so fundamentally different from open access or government effort regulations, they can be alternatives to MPAs. They can provide clear incentives for stock and ecosystem conservation at lower cost than imposed MPAs. The latter disrupt existing resource uses, raising the costs of implementing and enforcing MPAs (Costello and Kaffine, 2017). Multispecies fisheries, such as the Pacific groundfish fishery, can be included within quota or catch-share systems to broaden the impact of quota outcomes (Warlick et al., 2018). The literature finds that catch shares/quota are more successful with durable, secure, and tradable property rights. MPAs/EBM, however, potentially weaken property rights by imposing constraints on access and fishing activities.

As Holland (2018) argues, fishers can, through cooperation, often address external impacts more effectively and efficiently than can regulators if they are incentivised to do so. By capitalising on private information, they can devise restrictions on their own fishing that mitigate these issues more cost-effectively than by methods devised externally by agency officials (Little et al., 2016). Fishers organised around customary practices, formal catch shares, quotas, or TURFS can tailor contracts and incentives to the specific situation. Collaborating fishers can monitor each other more effectively and cheaply than can the regulator. They can adjust rules more quickly and in an incentive-compatible manner. They can discipline noncompliance relying on local pressure and norms, and they can invest in research to develop new technologies that reduce bycatch and habitat impacts.

As emphasised by Holland (2018), there is a general lack of involvement of fishers and other resource users in the initial stages of considering MPAs/EBM, especially where there is the possibility that the initiatives may not be implemented. The general pattern, instead, is for advocates to designate an MPA and then force those directly affected to respond. The absence of incentive-based, collaborative agreements is likely to be a major obstacle to achieving any reasonable response to ecosystem concerns (Dehens and Fanning, 2018).

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Output 2:

An Analysis of Ecosystem-based Management and Marine Protected Areas in New Zealand with Application to the Proposed Kermadec Ocean Sanctuary

Abstract

A review of New Zealand's experience with marine protected areas (MPAs) and ecosystem- based management (EBM) focuses on five factors: a) existence of clear and measurable ecological goals; b) incorporating both natural and social sciences in decision making and assessments; c) performance of rigorous trade-off analysis; d) involvement of Māori and other resource users; and e) incorporation of New Zealand's existing incentive-based management. Insights are drawn for assessing the proposed large-scale Kermadec Ocean Sanctuary that would be one of the world's largest MPAs with permanent restrictions on human access and use. The review concludes that emphasis has been on the mechanism rather than on measurable outcomes. Neither past nor proposed MPAs have had precise goals that address ecological interactions, uncertainties, timelines, or contingent adjustments. Claimed benefits, accordingly, are difficult to assess. Costs and their distribution are given insufficient attention to be able to determine their magnitude and the trade-offs encountered. The initiatives are driven by natural-science concerns without rigorous social science analysis. The absence of socio-economic investigation potentially undermines the achievement of environmental goals. Moreover, MPA efforts generally do not encourage the involvement of Māori or others whose knowledge and cultural values are critical in management of the ocean resource. The institutional framework of the quota management system (QMS) that has made New Zealand a world leader in fishery management has not been incorporated. Indeed, the large-scale Kermadec reserve could undermine the QMS and change user incentives across New Zealand's exclusive economic zone, resulting in greater resource depletion rather than protection. Finally, the Kermadec initiative may violate the provisions and spirit of the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992 and other Māori treaty rights.

Introduction

As a result of international agreements between parties to the Convention on Biological Diversity (CBD), there has been a growing pressure on country politicians and agencies to set aside between 10 percent and 30 percent of the world's oceans as marine protected areas (MPAs) and manage marine resources through ecosystem-based management (EBM) (CBD, 1996; CBD, 2004; CBD, 2010). While there are many databases documenting global MPAs, the *MPA Atlas* currently lists 14,688 individual MPAs. MPAs range in size from less than 1 square kilometre to 1,500,000 square kilometres and cover 10.2 percent of coastal and marine areas under national jurisdiction (Alcala, 1988; Barnett et al., 2016; Juffe-Bignoli et al., 2016). These protected areas can vary from strict nature preserves with very limited or no human use to protected areas where some highly regulated access to natural resources is permitted (Dudley, 2008). Ecosystem-based management is a broadly defined conservation strategy within MPAs that is motivated by the 'best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure, and function' (Christensen et al., 1996, 665).

MPAs and EBM are largely spearheaded by environmental non-governmental organisations (NGOs) and are increasingly promoted as ocean regulation tools (Sissenwine and Murawski 2004; World Wide Fund 2005; Halpern et al., 2010; Rand, 2017). Importantly, these efforts often are not initiated by national citizens who live and depend on the ocean resource and its long-term viability. Rather, they are presented as proving global public goods that may not be understood or valued by local users, whose actions are viewed by proponents as counter to the broad objective. Hence, MPAs and EBM involve centralised restrictions on entry and use, generally without user-group involvement. As such, the initiatives run counter to and do not take advantage of modern incentive-based fishery and ecological management arrangements that have documented successes (Costello et al., 2008; Branch, 2009; Walden et al., 2012; Birkenbach et al., 2017). Accordingly, there can be intense opposition, creating an adversarial setting. Depending on the political organisation and influence of user groups, opposition can corrode long-term citizen support for MPAs that is critical for securing ecological benefits. The work of 2009 Nobel Prize winner Elinor Ostrom (1990) has shown how critical it is to directly involve local agents in effective conservation of open-access natural resources. Documented resistance to large-scale MPAs is a paradoxical result for initiatives ostensibly designed to provide beneficial public goods, and it signals the need for more attention to be given to the trade-offs imposed and who bears them in MPA planning and implementation.

There is an overriding emphasis on the need to meet international treaty obligations for ocean habitat protection with little precision about what that means or how it will be achieved and sustained over an undefined period of time. MPA/EBM proposals emphasise ecological benefits with little socio-economic analyses of potential costs or trade-offs imposed on particular segments of the population. Such information is valuable because it directs MPA planning in ways that elicit the current and long-lasting support of important user groups and help in the selection of MPAs/EBM that are most viable. Instead, users are often portrayed as sources of ecological problems who must be constrained via imposition of controls, rather than as collaborators to achieve agreed objectives. Costs borne by such parties do not receive careful attention.

Social science is as important as natural science in achieving biological goals. A variety of issues can arise that rarely are addressed in planning. For example, what would happen politically within a country should the net costs of the initiatives rise? These include discovery of new very valuable fisheries or mineral deposits, while vaguely stated ecological benefits fail to materialise. What contingent adjustments would be considered should the MPA be found to be too small (or larger than required) or EBM is determined to be not restrictive enough (or more restrictive than necessary)? Budget allocations for scientific study, enforcement, management, and, potentially, compensation to injured parties are made at the country level. MPA/EBM efforts make little or no reference to the literature on the determinants and durability of country political commitments and budget appropriations in light of shifts in national costs and benefits.[1]

[1] For an example of the top-down push to establish and expand MPAs without consideration of local users and their incentives to cooperate and long-term national political factors, consider this recent statement by the CEO of WWF-NZ: https://thespinoff.co.nz/society/24-09-2018/why-nz-has-to-stop-telling-whoppers-about-our-care-for-the-ocean/

Further and importantly, achievement of the ecological goal is not independent of user reaction. The mechanism alone (MPA/EBM) does not insure success. Opposition raises enforcement and monitoring costs, and because enforcement will never be complete, protected habitat and species can be compromised. If user groups are sufficiently influential in the political process, international treaty provisions can be reneged or seriously weakened. Social science research in international collective action for global public goods reveals that, ultimately, countries take positions that are in their best interests should costs rise and be significant relative to the national benefits received (Barrett, 2007; Libecap, 2014).

New Zealand is a useful case study about the way MPA/EBM regimes have materialised through time and the potential trade-offs encountered. New Zealand is considered a pioneer of marine reserves, with the country's first reserve established in 1977 (Ballantine, 2014). As early as 1992, fisheries legislation called for an 'ecosystem based approach' to ensure sustainability (Wheeler et al., 1992). Although ostensibly created to provide national and global public goods, MPAs/EBM have been controversial, underscoring the importance of more consideration of socio-economic factors and national legal obligations. Research on the establishment of early reserves found support or opposition varied significantly according to how costs were imposed on communities (Wolfenden et al., 1994). Disagreements over whether, where, and how additional MPAs and EBM restrictions should be established continue (Bess and Rallapudi, 2007; Hale and Rude, 2017).

New Zealand is also useful because of its national application of incentive-based management through the quota management system (QMS) and the partnership between government and Māori in fisheries management decisions (Stokes, 1992). The QMS was instituted in 1986 and has grown from managing 26 species to 98 species and 642 individual stocks as of 2017 (Hale and Rude, 2017). Māori fisheries rights that were neglected for 140 years after the signing of the Treaty of Waitangi were formally incorporated into the QMS system in 1992. Māori received a 50 percent share in New Zealand's largest fishing company as well as 10 percent of pre-settlement quota and 20 percent of quota for any species brought into the QMS going forward (Treaty of Waitangi (Fisheries Claims) Settlement Act 1992). The expansion of the QMS has made New Zealand a global leader in incentive-based management and has given formal property rights to Māori that bind their interests in the sustainable use of fisheries – recognising practices that have long been part of their cultural resource-use ways. The QMS would provide a valuable institutional framework for additional habitat protection, should that be agreed on. It has not been central, however, to the MPA/EBM effort. In MPA planning, treaty obligations to Māori appear to be presented as secondary to the objectives of international agreements. There seemingly is little or no attempt to draw on the expertise of Māori and their enduring ties to a viable ocean resource or recognition of their legal rights to current and future New Zealand fisheries.

Methods

The analysis of MPAs/EBM in general is drawn from a comprehensive summary of the literature as listed in the references. The literature review is guided by common search terms, including ecosystem-based management, ecosystem approach to marine and fisheries management, marine protected areas, and marine reserves. Leading researchers in the area were also contacted to identify key studies.[2] The review of New Zealand MPA/EBM is organised around five categories: a) clarity of the ecological objectives with measurable benchmarks; b) integration of natural science and social science concepts and literatures; c) involvement of existing fishers and indigenous peoples in MPA planning, implementation, and management; d) extent of trade-off analysis and follow-up socio-economic impact assessment; and e) extent to which incentive-based fisheries management is incorporated in achieving ecological objectives.

To gain more in-depth understanding of how MPAs and EBM perform along these five categories, researchers spent three weeks in New Zealand interviewing stakeholders from Māori fishing groups, iwi, commercial fishing companies and industry groups, environmental non-governmental organisations, and research institutes. These interviews were semi-structured and guided in-depth research into New Zealand's experience with the global efforts for EBM and MPAs.

[2] Among those contacted were Luke Brander (University of Amsterdam), Chris Costello and Ben Halpern (UCSB), Lynne Zeitlin Hale (The Nature Conservancy), Dan Holland (University of Washington), and Dale Squires (UCSD).

Analysis of New Zealand's Experience with EBM and MPAs

New Zealand's Unique Institutional Context

New Zealand does not have law specifically designed to establish marine protected areas as might, for example, be defined under International Union for Conservation of Nature (IUCN) guidelines[3]. Rather, protection of marine areas in New Zealand has arisen through measures implemented under legislation established for a range of different management purposes as follows:

i. Forty-four marine reserves in New Zealand were created under the authority of the Marine Reserves Act 1971 although some are enacted through special legislation (e.g., Subantarctic Islands Marine Reserves Act 2014). The Marine Reserves Act 1971provides for the establishment of marine reserves for the specific and relatively narrow purpose 'of preserving, as marine reserves for the scientific study of marine life, areas of New Zealand that contain underwater scenery, natural features, or marine life, of such distinctive quality, or so typical, or beautiful, or unique, that their continued preservation is in the national interest'. The Department of Conservation is the principal implementing agency. All marine reserves are no take with the exception of Long Island – Kokomohua Marine Reserve, which allows traditional serpentine and nephrite harvesting.

ii. Eight marine mammal sanctuaries were established under the Marine Mammal Protection Act, 1978. Six of these place specific restrictions on fishing, primarily on location, trawling, and mesh size of set nets. In addition, general restrictions on trawling and set netting are also applied within most coastal areas around New Zealand.

iii. Ad hoc legislation applies in a number of cases and includes one marine park, the Mimiwhangata Marine Park (prohibiting commercial fishing but allowing recreational fishing) and the Sugar Loaf Islands Marine Protected Area Act 1991.

[3] The IUCN defined a protected area in its 2008 guidelines in this way: 'A 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 New Zealand Department of Conservation defines an MPA as 'An area of the marine environment especially dedicated to, or achieving, through adequate protection, the maintenance and/or recovery of biological diversity at the habitat and ecosystem level in a healthy functioning state.'

iv. A wide range of fisheries regulations established under the Fisheries Act 1996 for the purpose of providing 'for the utilisation of fisheries resources while ensuring sustainability' protect areas from fishing effects. These include, but are not limited to, a series of mātaitai reserves, established for management of traditional Māori fishing grounds, an extensive network of benthic protection areas, which prohibit fishing activity 100 metres from the sea floor, and seamount zones prohibiting all trawling around a range of these features. In addition, marine reserves, such as the Fiordland, Hikurangi, and the Subantarctic Islands marine reserves, have supporting legislative provisions coordinating the establishment of other types of management controls (such as fishing gear controls established under the Fisheries Act 1996).

v. Certain wildlife is protected under the Wildlife Act 1953 either specifically by species or within defined area delineated sanctuaries.

vi. A number of cable and pipeline protection zones, established under the Submarine Cables and Pipelines Protection Act 1996, prohibit fishing and anchoring as well as access in some cases.

vii. Finally, various coastal areas are protected under the provisions of coastal plans established under the Resource Management Act 1991 for the purpose of promoting 'the sustainable management of natural and physical resources'.

This interconnected set of measures protecting marine habitats and life that exist today has its genesis in the resource management reforms implemented during and immediately following the Fourth Labour Government free-market reforms of the 1980s. Prior to this, the management of what is now considered to be New Zealand's (government or Crown owned) conservation estate on land was administered by several large agencies for multipurpose use. The 1987 reforms abolished many of these agencies and divided state-owned land into land managed for commercial purposes, which were either sold or managed within state-owned entities, and conservation estate administered by a new single-purpose department, the Department of Conservation (Boston and Holland, 1987). A new Ministry for the Environment was established to take on responsibility for government environmental policy development. The many statutes regulating land use were ultimately consolidated into the Resource Management Act 1991, administered by local and unitary authorities, with a dedicated purpose as noted.

At the same time, the management of fisheries was placed under quota management in 1986. The quota management system, however, only partially privatised rights to fisheries by allocating individual tradeable catch quotas and retaining management of the marine sector of New Zealand within the authority of government under dedicated fisheries law. Initially, this new regime was implemented as an amendment to the planning-based Fisheries Act 1983, but it was later reformed into the Fisheries Act 1996 with a dedicated purpose of focusing on sustainable use. Importantly, this new legal framework was designed and implemented to meet settlement agreements on customary rights to fisheries reached with Māori, and a variety of mechanisms were put in place to manage interactions between fisheries law and other statutes. These mechanisms included, for example, provisions under the Resource Management Act 1991, which empowered local government to control land use (including use of the seabed), that exempted effects of fisheries harvesting from local body control. Likewise, the Resource Management Act 1992 and the Marine Reserves Act 1971 required explicit consideration of adverse effects on fisheries, and in the case of marine reserves, the Minister of Fisheries was required to agree to any such reserve before it could be put into effect.

For the above reasons, the New Zealand historical experience in marine management and establishment of areas protecting marine resources does not fit easily into MPA definitions, and the specific arrangements that might qualify as an MPA are often debated. Arguably, at one extreme, the QMS itself, encompassing all marine areas of New Zealand, could be defined as an MPA given that it is established under law to meet sustainability, including biodiversity maintenance and habitat protection. Recently, however, the New Zealand government has taken the position that a dedicated law is required to expand New Zealand's legal framework for MPA establishment given the narrow scope of the Marine Reserves Act 1971 (Ministry for the Environment, 2016b). This new legal framework is advocated on the basis that the current legal systems are not effective for managing New Zealand's marine environment. The Kermadec Ocean Sanctuary, proposed to encompass some 620,000 square kilometres of ocean space, although being promoted under dedicated law separate from the more general MPA legislation, is so far the most significant of these new MPA initiatives.

This analysis does not take sides in the debate about what constitutes an MPA but simply focuses on evaluating the process followed in establishing marine protected areas that are identified by the Department of Conservation in their 2018 Annual Report as New Zealand's current network of MPAs (which are limited to those established under the Marine Reserves Act 1971 and the Marine Mammals Protection Act 1978) and the more recent experience with the Kermadec Ocean Sanctuary process.

Lack of Measurable Ecological Criteria and Benchmarks

The literature exploring New Zealand's implementation of EBM and MPAs reveals an emphasis on the mechanisms themselves and an absence of measurable goals. Around forty-four MPAs were established in New Zealand under the Marine Reserves Act 1971. The Act's primary objective is to preserve special areas with distinctive features in their natural states for scientific study. The Act and subsequent planning documents do not identify metrics for monitoring and measurement of what is the natural state; whether or not an area remains within it; whether exogenous factors, such as ocean warming/salinisation, play a critical role; and what contingent adjustments might be made should the 'natural state' not be achieved by some unspecified time.

Despite a lack of clear, guiding, and generally agreed metrics for achieving or sustaining a natural state, there is evidence of monitoring in the reserves through to 2013, but this ceased in large part when government funding for such activities was reduced. At Cape Rodney-Okakari Point Marine Reserve (Goat Island), surveys have been conducted on fish and lobster abundance inside and outside the reserve. The reports on fish (Haggitt, 2011) and rock lobster (Haggitt and Freeman, 2014) point to increased size and abundance of species inside the reserve relative to the fished areas outside the reserve. Subsequent to 2013, monitoring activity has been limited mainly to the few reserves associated with university-based research.

Without determinate goals or benchmarks, and regular ongoing monitoring, it is difficult to assess whether reserves are providing wider benefits for management of the broader ocean resource. Is, for example, the goal to have larger lobsters in reserves while fishing and other activities are prohibited from these areas? Species depletion may rise along MPA boundaries and overall stock levels may not change as fishers compete for the reduced areas allowed for fishing. Other questions include how large should some fish stocks be? How does that impact predator/prey relationships and species richness? What is the likely impact on enforcement costs and political durability as protected species proliferate inside reserves but decline outside them?

MPAs in Australia and New Zealand saw lobsters increase in size and abundance inside reserves, but this was accompanied by decreases in urchins and abalone (Babcock et al., 1999). In California's Channel Islands, lobsters also increased in size inside reserves, but fishers experienced an estimated 28 percent loss in individual daily catches associated with loss of fishing grounds (Guenther et al., 2015). The costs of fisher search and adjustment, the time involved, and the present value of lost earnings during the process were not part of MPA planning or implementation analysis. Reimer and Haynie (2018) examined closures to protect Steller sea lions along Alaska's west coast and found significant economic costs on the groundfish fishery, some of which could be mitigated by shifts to new species and areas. Again, even in this programme evaluation, there was no assessment of the costs of search and adjustment.

Although the marine reserve system to date has been relatively limited, proposals for the dramatic increase of MPAs in New Zealand, such as the proposed Kermadec Ocean Sanctuary, change natural and socio-economic dynamics. The lack of measurable, transparent conservation objectives has made evaluation of past reserves a more subjective, rather than rigorous, scientific exercise. This condition has been tolerated within the country, but that may no longer be the case as the size and costs of MPAs/EBM rise. If user and broader political support for the reserves is to be secured and held for perpetuity, as plans describe, far more attention will be required for clearer benefit or outcome measures, timelines, tools for implementation, and possible contingent adjustments, along with greater assessment of current and future cost/benefit trade-offs to users and their communities.

Moreover, the more recent experience with marine protected area establishment in New Zealand in the case of the Kermadecs shows little consideration of coordination with existing fishery management. New Zealand is not unique in this case. Sanchirico et al. (2006) and Trochta et al. (2018) both explore the complexities and lack of agreement around how to operationalise EBM. In New Zealand, opinions on EBM range from calls for an overhaul of the QMS (Environmental Defence Society, 2016) to suggestions for a stepwise approach to incorporate the ecosystem in management schemes (Hilborn, 2004). Criticisms of the QMS or other incentive-based systems fail to outline the baseline alternative. Is it first-best conservation, and if so, what would that be and how would it be achieved? If second-best, how would EBM improve on existing QMS practices or how might QMS be modified to achieve collaborative goals? Yet, national discussions for implementation of EBM to meet international agendas do not seriously consider a bottom-up approach of collaboration with the QMS and building on its considerable successes. Without recognition of the value of existing practices, property rights, and treaty obligations associated with the QMS, fishers have little reason to endorse vaguely described EBM that may be unnecessary and costly and bring little overall additional habitat benefit.

Existing EBM discussions neglect ongoing management by government and fishers. In general, EBM is used to manage targeted species, reduce by-catch and minimise impacts on habitat (Hilborn, 2011). Much of these are already achieved in New Zealand. In an analysis of how thirty-three countries perform in terms of EBM principles and implementation, New Zealand was ranked third (Pitcher et al., 2009). Under the QMS, 84 percent of assessed stocks in 2018 were above management target levels (Ministry for Primary Industries, 2018). Fishers have also led efforts to reduce damage to sensitive ecosystems by setting aside roughly 30 percent of New Zealand's exclusive economic zone as benthic protection areas, where bottom trawling and dredging are prohibited to protect benthic habitat (Helson et al., 2010). Additionally, fishers and government have spent NZ$48 million on the innovative Precision Seafood Harvesting technology that selectively targets fish to reduce undersize catch and bycatch. The technology is projected to deliver NZ$44 million in economic benefits by 2025 (Guy and McKelvie, 2016).

The QMS and related fishery management also provide a framework for adaptive management. As quota holders with a long-term stake in the ocean resource, fishers can jointly incorporate new scientific data and respond with various management tools as is emphasised in successful EBM (McLeod et al., 2005). Fishers can do this because the quota system provides them with standing for collaboration not possible under open access, excessive entry, or traditional centralised fishery management. Under these systems, decisions are made by the regulator who may have different information and incentives, which raises monitoring and enforcement costs and reduces opportunities for success (Costello and Kaffine, 2017).

For example, the pāua (abalone) fishery has responded at a fine resolution to localised stock depletion and developed internal regulations to meet spatial differences in species. Pāua is unique as a sedentary and patchy species that requires certain densities to successfully spawn and is prone to localised serial depletion (Neubauer, 2017; Karpov et al., 2000). These circumstances are recognised by fishers, and the Paua Industry Council established a data-logger system that individual pāua divers wear to give industry and managers fine-scale, catch-per-unit effort data (Neubauer, 2017). This innovative, voluntary approach to fisheries data collection, promoted by the Paua Industry Council, has led to a spreading of harvest effort; deliberate increases in minimum harvest length, based on age of maturity of local populations; and voluntary catch reductions in areas that show signs of depletion (Jeremy Cooper, CEO Paua Industry Council, personal communication, August 2018). These adaptive and holistic initiatives under the QMS are clearly part of any EBM.

Objectives Are Driven by Natural Science with Limited Rigorous Social Science Analysis

Globally, proposals for marine reserves focus predominantly on natural science objectives (Thorpe et al., 2011). Social science, however, is as critical as natural science in achieving long-term ecological goals, especially if significant trade-offs are encountered. The neglect of socio-economic factors and the social science literature associated with them can ultimately undermine national efforts to promote enduring habitat and species protection (Christie, 2004).

In New Zealand, the primary purpose of the Marine Reserves Act is to hold areas in their natural state by excluding human interaction. The proposal for the Kermadec Ocean Sanctuary has an unambiguous objective to 'preserve the Kermadec region in its natural state now and in the future', and it outlines how fishing and other extractive activities will therefore be excluded from the area in perpetuity (Ministry for the Environment, 2016a). In regulatory impact statements, the government calls for monitoring of fish abundance, biodiversity, and habitat, with data to be made publicly available (Department of Conservation, 2011). Current and potential future impacts on resource users, their communities, and New Zealand citizens in general are not addressed in any detailed manner. There are no calls to monitor socio-economic indicators post-implementation as fish migration patterns change or as other exogenous factors change the importance of reserved areas to local populations. The ocean is heterogeneous and adjustment options vary and may be costly to implement. Prioritising natural science objectives and pristine environments that are only vaguely defined relegates human interests as secondary concerns.

This approach has serious implications for constituents who bear the costs of these proposals, the long-term political durability of support for the reserves, and the allocation of budgets for enforcement, monitoring, and management. Further, there is considerable empirical evidence from the work of Ostrom (1990) and others documenting the important role of local involvement in the conservation of natural resources. Without more attention to socio- economic issues, New Zealand could achieve apparent biological successes in some areas without clear benefits to important national stakeholders. This is observed in Southeast Asia (Christie, 2004) and in the lobster fishery around the Channel Islands in California (Guenther et al., 2015). An adversarial relationship between MPA advocates and the local population has been documented in many of the sanctuary efforts in Australia (Fitzsimons and Wescott, 2016) and the United States (Suman et al., 1999; Salz and Loomis, 2004; Hilborn, 2007; Stewart, 2009).

Finally, there has been little or no emphasis on MPA programme evaluation. Programme evaluation is a critical element of evidence-based policy making for natural resource and environmental management (Ferraro, 2009). Such evaluation assesses the degree to which changes in an outcome variable, such as species or habitat sustainability or growth, can be attributed solely to the MPA/EBM policy. It is essential for eliminating explanations for the outcomes that are unrelated to the policy or for signalling needed contingent adjustments. Most of the programme evaluation literature is devoted to establishing whether a programme causally affects an outcome variable. Programme evaluation also can explore the nature of the causal relationship (Imai et al., 2011). Understanding how a programme influences an outcome variable can assist people to design policies that achieve the intended objectives, adjust the mechanisms that do not, better balance costs and benefits, and elicit long-term constituent support (Ferraro and Hanauer, 2014).

Trade-off Analysis, Such As Cost-Benefit Analysis, Is Not Performed, and Impact Statements Lack Detailed Calculation of Likely Costs or Benefits

Standard cost-benefit analysis (CBA) of existing and proposed MPAs or EBM policies in New Zealand, surprisingly, is absent. Benefits claimed to arise from expanding marine protection, such as in the case of the Tāwharanui reserve, are based on broad assertions rather than measurement. These include generalised non-use gains from greater protection of the marine environment, tourism, education, other non-extractive recreation, and potential spillover for fishers (Department of Conservation, 2011). Additionally, in the impact analysis there is no recognition that any positive outcomes may accrue to different parties and communities – fishers are not necessarily trained in hospitality or tourism and long-term social/community dynamics of the two industries are quite different. There appears to be no sociological analysis of how communities might adapt to imposed shifts from fishing to tourism/recreation. Moreover, New Zealand citizens who bear no direct costs of the reserves will evaluate broad public-good benefits quite differently from those who do bear such costs. These consequences create distributional conflicts and political divisiveness within the citizenry, again potentially undermining the conservation goal.

In terms of cost, there is a similar lack of serious measurement, projection, or mitigation. For example, with the Tāwharanui reserve, proponents noted that there would be displacement of commercial and recreational fishers with little attention to the impacts on these parties and their communities. The reserve impact statement and other similar ones (Ministry for Primary Industries, 2015) rely on general claims of value to diffuse groups from additional protection. Search and adjustment costs receive little consideration, and regulators assume fishers can access fish in other locations at little loss, ignoring the heterogeneity of the ocean resource (Ministry for Primary Industries, 2015), or shift smoothly to tourism and recreation.

A post-hoc analysis of one, small (8.54 square kilometre) New Zealand marine reserve, Taputeranga, provides evidence that the costs can be significant. Rojas-Nazar et al. (2015) calculated the organisation and implementation costs to include NZ$508,000 for pre-establishment consultation and NZ$353,000 for post-establishment information dissemination, surveys, and equipment purchase. The authors point out that these are probably minimum estimates as much of the labour was voluntarily supplied by environmental non-governmental organisations, including the Royal Forest and Bird Protection Society of New Zealand and the South Coast Marine Reserve Coalition Trust. Management costs per area unit for this small reserve were similar to larger reserves – a finding that differed from previous projections (Balmford et al., 2004; McCrea-Strub et al., 2011). Estimates of annual management costs for the Taputeranga reserve range from NZ$43,200 to NZ$112,500. Additionally, after creation of the reserve, lobster fishers had higher search and travel costs than anticipated, with a total estimated annual displacement cost of NZ$22,160 per vessel (Rojas-Nazar et al., 2015).

Monitoring of the Taputeranga reserve after establishment found little evidence that there had been benefits to adult marine species within it or that spillovers outside reserve boundaries had occurred, with the exception of two species (Diaz-Guisado, 2014). Spillover-benefit claims are commonly made in reserve proposals, but the Taputeranga case and the Channel Islands case in the United States reveal that such projections may be optimistic and not forthcoming. More rigorous analysis of the factors underlying successful spillovers as well as the costs of search and adjustment by resource users are required prior to reserve establishment. Otherwise, planning and advocacy occur based on unproven and often subjective claims of generalised environmental and human benefits that can be achieved at low cost. If this turns out not to be the case, then overall reserve benefits and cost assertions are called into question. Negative reaction occurs when actual benefits and costs are revealed, harming prospects for existing and proposed reserves.

These results for a small reserve suggest caution for the proposed 620,000 square kilometre Kermadec Ocean Sanctuary, where costs and impacts on users may be much higher than the proposals suggest. Further, if there is widespread opposition, monitoring and enforcement costs could be higher still. In planning stages, careful trade-off analysis needs to be conducted using well-established cost-benefit analysis techniques (Hotelling, 1947; Clawson, 1959; Davis, 1963; Sorg and Loomis, 1984; OECD, 2006). This will make reserve proposals more realistic and point to areas where adjustments in reserve design and potential compensation are likely to be required. This process reduces uncertainty, makes the reserve process more legitimate, and encourages longer-term political commitment to the conservation objective.

National Calls for Expanding MPAs and EBM Generally Exclude Māori and Fishers' Perspectives

New Zealand follows a global trend of implementing MPAs and EBM in a centralised manner that does little to incorporate the perspectives and knowledge of indigenous peoples and other local fishers or the incentives relevant to them. In a global review of MPA literature, only 0.5 percent of papers dealing with MPAs included indigenous people (Ban and Frid, 2018). In New Zealand, there are distinct protocols that separate the government's process for reserves from ones implemented by Māori. Māori fishing interests are diverse across iwi (tribes) and hapū (clans/sub-tribes), and this heterogeneity influences Māori policies, as one would expect from locally supported approaches to conservation. Bess and Rallapudi (2007) discuss disputes arising from Māori holding commercial, recreational, and customary rights to fish in the implementation of taiāpure and mātaitai reserves/closures. Addressing different perspectives and expected costs and benefits is essential for a successful, collaborative approach to solving open-access resource problems (Ostrom, 1990; Ayres et al., 2018). These localised efforts to create taiāpure and mātaitai reserves and restrict access are quite distinct from government-led reserves, where there is less accommodation of diverse interests.

Reserve proposals in New Zealand have often neglected Māori positions in the planning process or rejected their stated preferences in implementation. For example, during the design of the Poor Knights Islands Marine Reserve, efforts were made to include local fishers and Māori. Local iwi strongly supported this approach, and the reserve plans restricted commercial and recreational fishing similarly. In the final adoption, however, the government reserve overlooked Māori goals and instead promoted recreational fishing (Guénette et al., 2000). More recently, with the Kermadec Ocean Sanctuary proposal, the government only consulted with two local iwi, even though other Māori hold fishery property rights in the area that would be closed (Ministry for the Environment, 2016a). Te Ohu Kaimoana, the Trust established through the Māori Fisheries Act 2004 to represent Māori fishing interests, criticised the government's push for MPA expansion without consulting Māori who own quota that would potentially be severely restricted with the loss of fishing grounds (Tuuta and Tuuta, 2018). These unilateral reserve actions taken by the New Zealand government are counter to the partnership between Māori and government that is articulated in the Treaty of Waitangi and reinforced in High Court decisions during the settlement process (Stokes, 1992).

The failure to genuinely include the positions of Māori and other fishers in reserve decisions undermines successful local collective action to support the conservation effort. The social science literature outlines conditions that facilitate cooperation among stakeholders for the provision of local and international public goods in natural resource management (Ostrom, 1990; Cox et al., 2010; Libecap, 2014; Ayres et al., 2018). MPAs are most likely to be successful over the long term if local users and their communities are directly engaged in the design and execution of reserves. This involvement insures that anticipated costs and benefits and their distributions are articulated and that rights are clearly recognised (Weigel et al., 2014). New Zealand, however, has a very mixed record in creating opportunities for such collective choice in natural resource management (Yandle, 2003). Even when government reserve legislation attempts to include indigenous peoples, actual co-management relationships fail to materialise due to the variety of Māori interests, a lack of trust in the process, and a lack of adjustment in positions taken by other interest groups (Taiepa et al., 1997). As they unfold, government reserve proposals become too uniform and too inflexible to accommodate the kinds of contingent adjustments required as new information emerges. Insights for flexible design and implementation are far more likely to be held by local users with a history of dependence on the ocean resource than by external reserve advocates, consultants, and government officials who have more remote and indirect knowledge.

EBM and MPAs Do Not Integrate the Incentive-based Fisheries Management Strategies Utilised in New Zealand

Proposals for MPAs and EBM fail to incorporate the incentives for habitat management and institutional structure created by the QMS. This is a critical missed opportunity because New Zealand is recognised as a world leader in incentive-based fishery organisation, and globally there is a considerable record of success for such arrangements. Catch-share programmes such as the QMS have eliminated the race to fish (Copes, 1986; Squires et al., 1998; Dewees, 1998; Birkenbach et al., 2017), improved fleet efficiency (Boyd and Dewees, 1992; Eero et al., 2005; Felthoven et al., 2009; Walden et al., 2012; Brinson and Thunberg, 2016), improved profitability (Dewees, 1998; Grafton et al., 2000; Campbell et al., 2000; Newell et al., 2005; Arnason, 2008; Costello et al., 2008; Bonzon et al., 2010; Essington et al., 2012; Waldo and Paulrud, 2013; Afflerbach et al., 2014; Thunberg et al., 2015; Birkenbach et al., 2017), and promoted ecosystem stewardship (Dewees, 1998; Squires et al., 1998; Campbell et al., 2000; Branch, 2009; Yagi et al., 2012).

A key advantage of catch-share programmes is that they provide stakeholders with an extended tie to the marine resource, depending on the characteristics of the rights granted. New Zealand has permanent fishing rights in the QMS, making them among the most secure and valuable in the world. Their value, however, depends on the vibrancy of fish stocks and the habitat that supports them. This is very different from an open-access setting where short-term considerations dominate. It is also different from centralised fishery management that is similar to MPA/EBM proposals, in which fishers have no internalisable stake in the regulatory process. Fishers are regulated entities not partners with standing to directly capture the gains from management and to bargain among themselves to adjust fishing and ecosystem-preservation practices that could increase those gains. These opportunities are possible under the QMS. Fishers benefit as rights holders from improved fish stocks and supportive ecosystems, whereas under centralised management, these benefits may or may not accrue to them, changing incentives to participate in collective resource management. These conditions underscore why catch-share systems have been so successful and why they could play a central role in additional marine conservation.

Moreover, the centralised, regulatory approach taken in proposing MPAs and calling for EBM undermines the QMS and the benefits it has provided. It has long been understood that the security provided in a property-rights system is a vital component for ensuring incentives are instilled in a rights-based system (Ostrom and Schlager, 1996). Exogenously imposed restrictions on access and use, previously available under the QMS, weaken confidence in the security of the rights granted and shorten time horizons, altering user motives for conservation. Further, placing MPAs/EBM on QMS stakeholders without their cooperation generates a sense of lack of legitimacy and fairness. This motivates resource users to create inventive ways to evade the restrictions, which compromises conservation objectives and raises monitoring and enforcement costs (Seabright, 1993; Wilson, 1995; Yandle and Dewees, 2003).

QMS critics claim that wider ecosystem impacts of fishing are ignored by quota holders (Slooten et al., 2017; Melnychuk et al., 2016; Whittaker et al., 2017), but these critics fail to provide a clear baseline for comparison. Is the baseline comparison with open access? With traditional fishery regulation? Or with proposed vague MPA/EBM regimes? Without more precision about how proposed EBM reforms would be implemented and maintained, relative to a clear baseline in a cost-effective manner, it is difficult to evaluate their merit. Moreover, critics fail to recognise the potential for use of the QMS as an institutional framework for further ecosystem protection, should that be an agreed objective.

Catch-share programmes elsewhere have provided an incentive-based arrangement for habitat protection. For instance, in British Columbia's groundfish fishery, which has been managed with catch shares since 1997, non-governmental organisations and fishers collaborated to create tradable quotas for sensitive benthic habitats such as sponges and corals (Wallace et al., 2015). After implementation, habitat damage declined to the lowest levels of harm in the seventeen-year data set (Wallace et al., 2015). Reimer and Haynie (2018) describe the way in which the Alaska groundfish fleet cooperative, within an annual total allowable catch, collaborated in the protection of Steller sea lions. Holland (2018) argues that fishers are best suited to address external impacts on non-target species and the broader ecosystem if they are incentivised to do so. Habitat quota programmes have been modelled to be more effective and efficient at protecting sessile non-target species than general MPAs (Holland and Schnier, 2006).

Within this overview of New Zealand's experience with MPAs and EBM, it is worthwhile examining a far more extensive MPA, the proposed Kermadec Ocean Sanctuary, to see how it is likely to perform.

Kermadec Ocean Sanctuary Case Study

The projected Kermadec Ocean Sanctuary illustrates the problems associated with the MPA and EBM process in New Zealand and that the proposal could have far more serious consequences because of its size and broader implications. The proposal, for what would be one of the world's largest MPAs (620,000 square kilometres or 15 percent of New Zealand's exclusive economic zone), signifies a major shift in New Zealand policy from relatively small reserves for scientific research to a large-scale marine protected area for general global conservation objectives. Analysing this proposal and the repercussions in detail highlights the challenges it poses to Māori rights, culture, and livelihoods, as well as to the overall New Zealand QMS. It also shows a lack of information about additional conservation and socio-economic goals the sanctuary would causally achieve.

In planning documents for the sanctuary, the government does not provide evidence of threats to the area or any measurable conservation or socio-economic goals. In the impact statement, the government outlines global threats to marine environments from overfishing and climate change, while at the same time emphasising the unspoiled nature of the Kermadec area and the few immediate and direct threats to the region (Ministry for the Environment, 2016a). The emphasis is on preserving this vast area as an unexploited ocean reserve. There are, however, no indicators or targets for species richness, biodiversity, fish abundance, habitat quality, or other ecological factors to be protected. Also, it does not specify what the causal mechanisms might be beyond a no-use designation. What other factors might affect the species and habitat in the region over time in light of dynamic ocean and climatic conditions? The planning documents provide a list of selected species and ecological features of the Kermadec area, but there is no indication that these are at risk or are depleted or whether the only goal is to retain their current state. The Kermadec initiative appears to set aside a very large area of the ocean resource in its present state by fiat with no clear outline of conservation goals, careful assessment of benefits and costs and who might bear them, timelines, or programme evaluation. Nor does it outline contingent adjustments that would be considered should conditions deteriorate for exogenous, unanticipated reasons or if QMS fish stock migration patterns were to move into the region. If the initiative was small, then these issues might be of little consequence, but it is not, and potential longer-term costs could undercut national efforts to conserve truly threatened areas of the marine environment.

The sanctuary proposal is driven by international non-governmental organisations and government officials wanting to be recognised as leaders in the MPA movement. The planning document lists major advocacy parties, including Pew, World Wide Fund for Nature, and the Royal Forest and Bird Protection Society, and notes the desire for New Zealand to be viewed as 'at the forefront of global protection initiatives' (Ministry for the Environment, 2016a, 5).

Although the government impact statement describes general costs and benefits of the proposed sanctuary, it does not provide a rigorous cost-benefit analysis, which is justified given the magnitude of the initiative. It lacks the fundamental components of cost-benefit analysis that could describe the possible socio-economic effects and the trade-offs that might be imposed. For example, there is no discussion of uncertainty in achieving ecological benefits or factors that might affect it; no clear timeline for cost assessment on the commercial fishery nor analysis of the opportunity costs and greater search costs arising from denying access to so large an area; and no use of discount rates for assessing costs and benefits over time. The impact statement outlines current fishing interests in the region and estimates total economic value to be NZ$164,672 (Ministry of the Environment, 2016a, 8). This is framed as a small fraction of the value of all fisheries in New Zealand, suggesting that harvests could occur elsewhere by the same quota holders. Such an assertion ignores the heterogeneity of the ocean, future shifts in fish stocks, and the costs of search and learning that are borne by quota holders. These negative outcomes were neglected in the United States Channel Islands reserve proposals, but significant, uncompensated costs were imposed on fishers as the reserves were put into place.

The Kermadec impact statement acknowledges opportunity costs associated with locking up an area in perpetuity but makes no attempt to calculate these because ‘it is difficult to quantify this opportunity cost’ (Ministry of the Environment, 2016a, 8). Difficulty, of course, is no justification for a lack of analysis. A failure to measure opportunity costs implicitly assumes they are minimal, which they may not be. As noted earlier, there is established social science research in cost-benefit analysis to estimate trade-offs imposed by policy and the distributional impacts on the parties that actually incur the costs or receive the benefits. These distributional effects critically affect the success of resource management regimes (Ostrom, 1990). None of this analysis is evident in the report. Vaguely described, globally broad benefits are asserted without dimension or identification of causal mechanisms, and costs are effectively dismissed. Compensation is directly ruled out. It is worth noting that in a similarly large expansion of marine reserves in Australia’s Great Barrier Reef Marine Park, fishers were expected to incur serious losses in access in favour of broad ecosystem and tourism benefits and were compensated through the buyback of licences and funding of transitional programmes (McCook et al., 2010; Macintosh et al., 2010).

Such an incomplete causality and cost assessment by proponents of the Kermadec Sanctuary and reliance on unspecified, diffused benefits generates mistrust and weakens the cooperation needed with resource users that research reveals is essential for successful conservation. It creates social divisiveness and sets the stage for political revision within the country at a later date should costs rise and commensurate benefits not be apparent. Moreover, as argued above, the arbitrary set aside of a large portion of New Zealand’s exclusive economic zone with potential fishery opportunities challenges the strength of the property rights granted under the QMS that have made New Zealand a world leader in fishery management.

Māori fishing rights are briefly addressed, but the reserve’s impact on them is given little attention in the sanctuary proposal (Ministry for the Environment, 2016a, 8–9). As mentioned earlier, the vast majority of MPA literature, including that associated with large-scale MPAs, does not include indigenous people in planning efforts (Ban and Frid, 2018), which alienates these groups and fails to take advantage of their localised knowledge of the resource and how to manage it (Leenhardt et al., 2013). The Kermadec impact statement acknowledges that there is a risk that the imposed sanctuary would be perceived as undermining the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992 (Ministry for the Environment, 2016a, 9). Indeed, Te Ohu Kaimoana has taken the government to court over the initiative, and one of the two iwi consulted on the proposal withdrew support (Bootham, 2016).

The quota held by Māori around the Kermadec Islands would be compromised by the access and use restrictions included in the reserve without financial compensation. Such compensation would not only include estimated value of lost access, but any reduction in overall quota values arising from the imposition of the sanctuary. The magnitude of the set aside and its precedent creates uncertainty for QMS property rights in general, undermining their role in promoting sound fishery management throughout New Zealand waters and internationally, given the highly migratory status of many fish stocks around the Kermadec Islands. This broad effect also is neglected in the proposal. Finally, the absence of extensive consultation, cooperation, and collaboration with Māori in the sanctuary proposal neglects Māori cultural attachments to the resource and understandings of how to preserve it. Considerable research into Māori perspectives of kaitiakitanga, often translated as guardianship, points out the deeply held values around sustainable, wise use (Kawharu, 2000; Roberts et al., 1995) that are essential for resource conservation. Such local knowledge and attachment is not part of the background of MPA advocates.

Conclusions

The proposed large-scale Kermadec Ocean Sanctuary, with its greater implications for fishery and marine management, signals a fundamental shift from the previous, relatively modest establishment of MPAs with related EBM in New Zealand. The approach taken may jeopardise past conservation and social gains associated with New Zealand's ocean policies and those promoted with the sanctuary. Assessing how this large-scale MPA has been proposed and implemented at the national level suggests caution is needed as New Zealand moves forward to meet general ecological concerns. This review focuses on five areas of concern: a) existence of clear and measurable ecological goals; b) incorporation of both natural and social sciences in decision making and assessments; c) performance of rigorous trade-off analysis; d) involvement of Māori and other resource users; and e) incorporation of New Zealand's existing incentive-based management into proposals for MPAs and EBM.

In general, New Zealand's MPAs/EBM have neglected social science methods and analysis to appropriately propose and assess socio-economic impacts and how they in turn could affect achievement of conservation objectives. With limited, small-scale reserves, these effects may be of little consequence, but with large-scale ones, such as the Kermadec Ocean Sanctuary, the impacts are likely to be far more significant and important for marine policy.

Proposals for MPAs and EBM focus on broad environmental objectives that are motivated by international agreements, non-governmental organisations, and national political officials who seek to have New Zealand be a leader in ocean conservation. The initiative planning and implementation documents generally do not describe costs or benefits with any precision, identify sources and effects of uncertainty in achieving ecological goals or the related costs of doing so, or determine timelines or discount rates. Additionally, advocates do not personally bear the socio-economic costs of their actions. There is little attention to programme evaluation in planning or implementation, and underlying causal mechanisms between establishment of the sanctuary and claimed outcomes remain unclear.

Neglect of trade-offs does not mean that they do not exist, and national negative political reaction is likely should benefits be perceived as limited and costs high (Libecap, 2014). Moreover, the parties that do bear direct costs are unlikely to cooperate in achieving ecological objectives that are often framed as broad global public goods. Failure to generate cooperation with Māori, who have cultural ties to resources, and with other QMS holders who have a stake in the ocean resource could compromise success.

These problems are not unique to New Zealand as there is an understanding that clearly defined goals, causality linkages, and testing of assumptions are critical for MPA success worldwide (Agardy et al., 2003). Current monitoring of New Zealand's reserves emphasises increases in abundance and size of some species (Haggitt, 2011; Haggitt and Freeman, 2014), but it does not address the impacts on prey species that have occurred in other places or impacts on fishers who may not have seen the benefits from spillovers that proponents had asserted (Christie, 2004; Guenther et al., 2015). In planning for new and larger reserves, the ecological and social goals should be clearly stated in a testable fashion so the inherent trade-offs can be evaluated rigorously along with the overall programmatic performance.

This review reveals a lack of involvement of Māori and fishers in the MPA and EBM process. New Zealand's broad application of rights-based systems has helped the country become one of the world's leaders in avoiding overexploitation of fish stocks (Beddington et al., 2007; Worm et al., 2009). Rights to fish were a fundamental part of compensating Māori for over a century of violations of the Treaty of Waitangi. The brief attention to these rights in the proposal for the Kermadec Ocean Sanctuary raises a number of concerns. First, it appears to violate the objectives of the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992. Second, actions taken by the government that erode the security of fishing rights could have ripple effects in the broader fisheries management regime, undermining existing incentives for marine stewardship and eventually creating the exact environmental and social problems that MPAs and EBM are designed to avoid. Third, Māori and other resource users need to be involved in collaborating on solutions, rather than being cast as adversaries, to draw on their unique, local, long-standing understanding of the resource and how to protect it. This was a key insight in Elinor Ostrom's 2009 Nobel Memorial Prize in Economic Sciences that has been missed in the effort to set aside large resource areas. MPA/EBM efforts are motivated by natural science concerns, but without careful social science evaluation and the collaboration of the people whose knowledge and support are critical across time and across shifting political cycles, the ecological objectives are unlikely to be obtained.

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Output 3:

Reallocation to Open-access Recreational Fishing: An Examination of the Impact on New Zealand's Quota Management System

Abstract

When commercial and sports fisheries access the same stock but are differentially regulated, fish populations can be imperilled and the viability of the most constrained regulatory system compromised. Worldwide, sports/recreational fishing is expanding, often targeting fish stocks exploited by commercial fleets. Moreover, for a variety of reasons, sports fishing is less constrained than commercial harvests and is controlled as regulated open access. In developed countries, citizens' access to fishing areas is viewed as a right, and limited regulations focus on effort controls. These controls, however, lack the incentive effects of rights-based management used in many commercial fisheries. Measurement and monitoring are limited. Overall, the growth of the sports fishery results in greater entry and negative effects on the stock. This, in turn, undermines the basis for rights-based management and its documented successes. In New Zealand, greater allocation of harvest to sports fishing potentially lowers values in the quota management system (QMS) and weakens the property rights granted to Māori in the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

Introduction

Recreational fishing, both from for-hire charter vessels and by individual sports anglers, is an important and growing activity worldwide. It accounts for perhaps 12 percent of global fish harvest (Abbott, 2015, 1). This growth coincides with rising per capita incomes, reduced transportation cost to the most lucrative locations, and ease of entry by individual citizens. In most countries, access to marine resources by recreational fishers is viewed as a right of citizenship, and there is little effort to constrain entry. There are strong political and practical pressures to accommodate recreational fisheries when so many citizens are potential participants. Recreational fishing provides leisure and sources of protein, and it supports local fishing communities. Nonetheless, the associated increased fishing effort can deplete fish stocks and undermine long-term sustainability. In the United States, for example, an estimated 9.6 million recreational fishers engaged in 16 million trips annually, and in the early 2000s, they contributed 23 percent of landings among marine fish populations that were overfished or experiencing overfishing (Abbott et al., 2018, 8948).

In developed countries, recreational fishing is loosely managed through regulated open access with a variety of effort controls, including adjustable fishing seasons, bag limits, and size restrictions. Entry typically is open to all citizens. These regulations encourage a race to fish early in the season when stocks may be concentrated and before congestion sets in or managers reduce season length or adjust bag limits or acceptable fish sizes. Due to the diverse nature of sports/ recreational fisheries in terms of fisher numbers, ports, vessel types, and harvest practices, measurement and monitoring of the impact on stock mortality (landings and discards) is limited, certainly relative to commercial fisheries (Abbott, 2015, 3).

This setting contributes to overharvest and stock declines: the race to fish encourages excess investment in vessels, equipment, and labour, which raises costs; and a derby fishery limits the ability of fishers to spread their fishing spatially or temporally in a manner that would generate the most value. As harvest pressures rise, seasons are narrowed and congestion increases. In the United States Gulf of Mexico red snapper sports fishery, for example, the June 2014 season was lowered to nine days (Abbott et al., 2018, 8949). Based on survey data, Abbott et al. (2018, 8951–8952) estimate that some type of second-best management system with tradable vessel days, the assignment of a fixed number of annual fish tags, or angler management cooperatives (Abbott, 2015, 13–15) would generate important welfare and stock gains over regulated open access. Depending on the system adopted, discard rates could fall by 40 percent, enhancing the stock and lowering fishing costs, and the benefits to the recreational fishery could be US$1.2 billion in the United States alone, and US$30 billion worldwide, (Abbott et al., 2018, 8952).

These estimates of the gains to the recreational/ sports fishery from improved management are certainly an underestimate of the overall welfare benefits. Whenever a sports fishery accesses the same stock as a commercial fishery, there is overharvest in the former and associated fish stock depletion and ecosystem deterioration that undermine value in the latter. This dynamic intensifies the race to fish in both fisheries, generating additional losses in resource rents. Where the commercial fishery is governed by a total allowable catch with associated catch shares or quotas, over entry and excessive harvest in the sports fishery reduces the value of individual quotas and property rights in the commercial fishery. Losses in quota value reflect deterioration in current and future fish stocks as well as uncertainty for individual quota owners who can no longer predict that their changes in fishing practices will result in them capturing a larger flow of resource rents.

These inter-sectoral competitive losses are potentially significant. Indeed, the patterns of higher cost, reduced profits, and stock depletion in the recreational fishery have been encountered historically in commercial fisheries (Grafton et al., 2000). In those fisheries where rights-based quotas have been implemented, there have been dramatic gains. Considerable research has documented the environmental benefits of quota systems with their ability to eliminate the race to fish (Birkenbach et al., 2017), reduce the likelihood of stock collapses (Costello et al., 2008), and generally promote environmental stewardship (Branch, 2009; Yagi et al., 2012). Research has also demonstrated that rights-based fisheries management has increased fleet efficiency and profitability (Arnason, 2008; Felthoven et al., 2009; Walden et al., 2012; Essington et al., 2012; Thunberg et al., 2015; Brinson and Thunberg, 2016; Birkenbach et al., 2017).

The key drivers in quota systems are change in incentives and time frames and the ability of quota owners to contract among themselves to improve conditions. The motives to race to fish and to invest in excessive capital and labour are changed, and profits rise. Fishers can harvest over longer periods, knowing that entry is limited, and invest in higher-valued product (Grafton et al., 2000). These improvements are incorporated in rising quota values (Newell, Sanchirico, and Kerr, 2005) that are captured by quota owners as property rights. Inter-sectoral fishing competition for the same stock under different management regimes, however, can reduce quota prices and the value of the property rights associated with them.

Despite the evidence of fishery gains from rights-based systems, sports fishers have resisted them due to feared constraints on access and harvest. As noted above, sports and recreational fishers are not only numerous but heterogeneous, with multiple techniques, locations, and boat and equipment types, including small, individual vessels and larger for-hire boats. Compared to many commercial fisheries where numbers are smaller and vessels larger and more homogeneous, sports/recreational fisheries are more diverse, which makes collective action and forging a clear management position far more problematic. Moreover, because of limited regulation and open access, there is less information regarding the way sports fishing contributes to overall stock declines, compared to commercial harvests, and how benefits from stricter controls would accrue to individual fishers.

Sports and Commercial Fishing Competition in New Zealand

New Zealand is considered an international leader in rights-based fisheries management, with one of the most widely applied quota systems in the world (Lock and Leslie, 2007). The quota management system (QMS), guided by the Fisheries Act 1996, currently manages 642 fish stocks consisting of 98 species (Ministry for Primary Industries, 2018). The QMS ostensibly regulates commercial, recreational, and customary (Māori) fishing as three distinct sectors, with a total allowable catch (TAC) for each stock split among these user groups. Under this system, New Zealand has been uniquely able to act pre-emptively and avoid overexploitation of fish stocks (Worm et al., 2009). Currently, nearly 80 percent of stocks in New Zealand are managed at or above their target level (Ministry for Primary Industries, 2018).

A growing recreational fishing sector, however, could threaten the stability of the rights- based system in place in New Zealand, and it is particularly problematic for Māori quota holders. During the Deed of Settlement 1992, Māori were granted $150 million to purchase half of Sealord, New Zealand's largest fishing company, and quota within the QMS (Treaty of Waitangi (Fisheries Claims) Settlement Act 1992; Hale and Rude, 2017). Māori received 10 percent of all quota in the QMS as of 1989 plus 20 percent for new species brought into the system after 1992. As of 2016, these holdings were valued at more than NZ$1.4 billion (*Stuff*, 2016). At the same time, the recreational fishery in New Zealand has expanded, and in many cases it accesses the same stocks covered by the QMS. New entry and fishing pressure in the recreational sector lowers fish stocks and potentially undercuts quota values and the performance of the QMS. Quota holders have less certainty about fish abundance and the benefits of their rights-based harvest practices; as more returns are captured by recreational fishers, quota holders may experience catch reductions and shifts in the time of harvest. The outcomes, as noted above, are lower quota prices and values of the property rights associated with them. Depending on the size of the inter-sectoral competition, the successful QMS could be placed at risk if the value of participation is lowered. This effect, in turn, directly affects Māori and the provisions that they were granted under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

Methods

This paper assesses how the recreational sector has fared in New Zealand and how it potentially impacts the property rights held by quota holders, including Māori who are covered by provisions in the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992. To accomplish this, an in-depth literature review was conducted, focusing broadly on fishing sectors and allocation in New Zealand, and more specifically on two important stocks as case studies: Snapper 7 (SNA7) and southern bluefin tuna (STN1). These two stocks have seen increases in recreational participation, and Māori and the wider industry have been concerned over dilution of their property rights. These case studies also cover distinct types of recreational/sports fishing, with bluefin tuna being offshore large-game fishing and snapper being New Zealand's most popular inshore fishery.

Literature reviewed focuses on government documents explaining allocation and scientific analysis of these fisheries. The analysis in this paper also benefits from assessing relevant natural and social science peer-reviewed literature covering fishing in New Zealand and the fundamental components of rights-based fishery management.

To gain a more in-depth understanding of how fishing allocation effectively has shifted in response to recreational fishing demands, researchers spent three weeks in New Zealand interviewing stakeholders from Māori fishing groups, iwi, commercial fishing companies and industry groups, environmental non-governmental organisations, and research institutes. These interviews were semi-structured and guided in-depth research into New Zealand's experience with shared fisheries. The interviews focused on the concerns of stakeholder groups, as well as the government's response to addressing multi-sector fisheries. Interviews supplemented the literature reviews and provided insight into topics that have largely not been addressed in scientific literature or government documents and highlighted issues raised in the two case studies.

Examination of Shared Fisheries in a Rights-based System and Potential Impacts on Māori Property Rights

The gradual reallocation of access to fish stocks to recreational/sports fishers challenges the essential incentives for the quota management system. Elinor Ostrom's seminal work (1990) on managing common-pool resources stressed the need to clearly define who can use resources and also the importance of sufficient monitoring of these users' behaviour. Empirical analysis strongly supports her claims that these two principles are important for sustainable management (Cox et al., 2010).

New Zealand's recreational fishing, using small individual vessels or larger charter boats, is essentially managed as open access, with entry allowed for all citizens without fishing licences or formularised reporting of catch. Regulators rely on effort restrictions, including area restrictions or closures, and daily bag and size limits along with surveys to estimate catch (Wynee-Jones et al., 2014; Fisheries New Zealand, 2019). There are seven marine fishing areas with generally similar rules.

Effort controls and regulated open access lack economic or ecological incentives for stakeholders to behave consistently with management goals (Hilborn et al., 2005), and this explains why rights-based systems in commercial fisheries have had success. There is no reason to expect that incentives and outcomes for effort controls would be different in the growing recreational sector. With so many possible entrants in many different ways and voluntary catch reporting, monitoring and measurement of total harvest and information about the impact on the stock are very incomplete. Given the number of participants, the effects could be large and increasing. For example, the survey process that managers rely on to monitor the recreational sector estimated a 2.8-fold increase in snapper harvest from 1996 to 2000 but projected that the number of households with recreational fishers increased from 13.9 percent to 51.4 percent over the same time period (Kearney et al., 2012). In light of these disparities between estimates of entry and catch, it seems likely that the harvest growth was significantly underestimated.

In New Zealand, the recreational sector has become a significant factor in many important fisheries across the seven regions regulated by the government. The large number of current and possible participants creates a formidable political constituency that few politicians or regulators can ignore. As a result, the sector has received effective increases in fish stocks, at the expense of the commercial fishery (Hale and Rude, 2017). If, however, harvest in the sports fishery is greater than regulators believe, as is likely, then total harvest across both sectors could rise, imperilling stocks. Reductions in available total allowable catch for quota holders and declining stocks potentially reduce QMS values. The recreational sector has resisted either licensing that could limit entry or formal inclusion into the QMS. This opposition is understandable from the point of view of individual sports fishers because access historically has been free and open (Borch, 2010; Council of Outdoor Recreational Associations of New Zealand, 2017). The situation, however, is not sustainable over the long term. It would not be sustainable for fish stocks or values to vessel owners or their customers even if the commercial fishery was totally banned from specific areas. The current arrangement lacks needed incentives and information for maintaining values and fish populations across time.

Alternative, second-best controls, noted above, could provide some harvest restrictions and shifts in incentives beyond current, limited open-access effort controls that have failed in other settings.

Explored below are two New Zealand case studies that illustrate the growing problems from inter-sectoral rivalry for the same fish stocks. Comparisons are drawn from the United States' quota system for the Gulf of Mexico red snapper fishery. The two case studies, Snapper 7 and Southern Bluefin Tuna 1, are of particular concern because of the implications for Māori QMS stakeholders.

Snapper 7 (SNA7)

Snapper is the most valuable inshore species in New Zealand. Between 2010 and 2015, the average value of commercially landed snapper was $61 million (Williams et al., 2017). Values of the fishery vary by method of estimation. Using a marginal willingness-to-pay approach, the snapper fishery is estimated to be $15.8 million annually, while an average willingness-to-pay approach estimates the fishery to be worth $85.1 million (The South Australian Center for Economic Studies, 1999). In either case, the fishery is attractive and of growing interest to both recreational and commercial fishers.

Snapper 7 is a good example of how conflicts over allocation of shared fisheries have played out. This stock covers the top and west coast of the South Island of New Zealand, and it includes Nelson and the Marlborough Sounds, which is the largest seafood region in New Zealand (Pavlovich and Akoorie, 2010). Like many of the snapper stocks in New Zealand, SNA7 has experienced considerable variation in health and landings. A recorded low in landings occurred in the 2001 2002 season at 141 tonnes. Since 2009, the fishery has recovered as the stock has rebounded (Ministy for Primary Industries, 2017; Langely, 2018). Allocation to the commercial sector has remained stable at 200 tonnes annually from 1997 to 2014. Of concern to QMS quota holders is how much of the stock will be effectively assigned to the less-regulated recreational sector, affecting quota values (Dewees, 1998, S135).

In 2016, the Ministry for Primary Industries increased the recreational allocation from 90 tonnes the year before (of which they estimated only 83 tonnes were caught) to 250 tonnes for the coming year (Ministry for Primary Industries, 2016). This near tripling of the allotted catch to the recreational sector was based on a proportional increase in harvest, relative to biomass. The ministry predicted that if biomass of a stock increased threefold, then recreational catch could triple as well (MPI, 2016, 5). Under the new allocation, the previous 70/30 split between the commercial and recreational sectors shifted to 50/50 for SNA7. The Southern Inshore Fisheries Management Company, an established Commercial Stakeholder Organisation, challenged the science underlying the increase in authorised recreational catch (Southern Inshore Fisheries, 2016, 22). Other stakeholders have joined in the criticism. Without reliable data on biomass, sports-sector harvests, and stock effects, the greater allocation could impact fish populations. Subsequent major reductions in allocation to the recreational industry are politically difficult, suggesting that existing regulation would not adequately respond to new stock conditions.

Māori fishing rights, formally recognised in 1992, are affected by the allocation shift from the commercial sector to recreational fishers. It dilutes Māori access to improved fish populations, and if weak regulation of the recreational sector leads to overfishing of the shared stock, Māori quota values are at risk. This policy-induced result can be viewed as a violation of the Deed of Settlement 1992. The Ministry for Primary Industries called for input from Māori stakeholders (MPI, 2016, 3), but the reallocation was unilateral. It is difficult to see how Māori as commercial quota holders would benefit from a major increase in distribution to the recreational sector.

Southern Bluefin Tuna (STN 1)

New Zealand is a member of the Commission for the Conservation of Southern Bluefin Tuna (CCSBT) that allocates harvest among member states. New Zealand regulators manage their national allocation of tuna as a single stock, STN1. This stock, like others, is split among commercial, recreational, and customary Māori sectors. STN1 were brought into the QMS after the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992, which granted Māori 20 percent of the initial quota. From 2018 to 2020, due to better forecasts of southern bluefin tuna stocks globally, New Zealand was allocated more tuna from the CCSBT (Commission for the Conservation of Southern Bluefin Tuna, n.d.). Eighty-eight tonnes of additional bluefin tuna were split by the New Zealand Minister of Fisheries: 76 tonnes to the commercial fishery and 12 tonnes to the recreational fishery, which increased the latter's share of the overall bluefin tuna fishery total allowable catch (Nash, 2018). This action, however, reduced the portion going to Māori, continuing a trend from the 1990s (Tuuta, 2018).

As with Snapper 7, Māori gradually lose access to quota that was guaranteed under the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992. Moreover, as discussed earlier, the recreational fishery has fewer effective controls, threatening tuna stocks that the QMS and other quota systems are designed to protect. Indeed, recreational fishers celebrated the 'year of the tuna' in 2017, when the new allocations were announced, with reports of large numbers of sports fishers targeting bluefin around Waihau Bay and Gisborne every weekend (*The Adventurer*, 2017). With ease of entry and limited monitoring or measurement, it is possible for the recreational sector to quickly overshoot its allocation. The rapid expansion of the sports bluefin tuna fishery was not matched by the infrastructure to support it, resulting in waste. This would probably not occur under Māori stewardship where there is a tradition of wise use and respect for a prized fish caught (Peter van Kampen, personal communication, 2018).[4]

Gulf of Mexico Red Snapper

The inter-sectoral conflicts that arise in New Zealand when different segments are regulated in dissimilar ways are not unique. The Gulf of Mexico red snapper fishery in the United States has experienced many of the same issues that illustrate the problems facing Māori and other QMS quota owners. As with shared fisheries in New Zealand, recreational fishers in the Gulf of Mexico take a significant portion of the total catch of red snapper (Colemen et al., 2004). In response to derby-style fishing, falling fish populations, and reduced profits, an annual total allowable catch limit was implemented in the snapper fishery that split harvest roughly equally between the commercial and recreational sectors, with the recreational sector further divided into for-hire charter boats and private anglers. In 2007, an individual fishing quota system was implemented for the commercial fishery (Weninger, 2008; Agar et al., 2014).

Under these arrangements, regulators in federal waters have limited the total number of fish to be harvested annually, restricted the number of fishing licences issued for commercial and recreational for-hire vessels (but not individual anglers), reduced the number of fish retained per trip, added minimum fish-size limits, restricted gear types, and set fishing seasons. Although stock assessments indicate that red snapper abundance has risen since 2007, greater catch-per-unit of effort and more entry in the recreational fishery has led to shorter sports-fishing seasons. Even so, that sector has exceeded its annual quota. Moreover, Gulf States have opened state waters to recreational red snapper harvest for extended periods when federal waters were closed.

Overall, the Gulf of Mexico management system has had mixed success since 2007. In the first five years, profitability and resource stewardship metrics showed significant gains in the fishery, but disputes over distribution of those gains remained (Agar et al., 2014).

Because of the close parallels with New Zealand, the experience in the fishery is useful. As in New Zealand, the recreational sector, particularly the individual angler subsector, has fewer constraints. It has exceeded its allocation; experienced shorter allowed fishing seasons; and been embroiled in lawsuits with the commercial sector, between state and federal regulators, and between the two recreational classes of anglers (Environmental Defense Fund, 2017; Pew, 2016). In neither the Gulf of Mexico red snapper fisheries nor the SNA7 and STN1 cases are recreational fishers incorporated into the quota systems in a significant way. Because of increased entry and overharvest in the Gulf of Mexico, individual anglers have faced tighter season limits (at one point just three days

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in federal waters), a continued race to fish, and problems with state and angler compliance with federal regulations. At the same time, overfishing by the recreational sector weakens the quota system put in place for the commercial fishery. As commercial fishers reduce catch to build stock and increase profits, they observe rising harvests by competing sports fishers.

Conclusions

New Zealand's quota management system is heralded internationally as one of the most progressive and comprehensive fisheries management regimes. This approach relies on incentivising rights holders to incorporate long-term sustainability into fishing decisions since rights holders will be able to capitalise on fish stocks into the future. A growing recreational sector that is managed as regulated open access does not internalise these same incentives. Successful management regimes, from both an economic and ecological standpoint, require institutionalised systems that create such incentives for stakeholders across sectors to behave in a manner that promotes conservation goals (Hilborn et al., 2005). Reallocating quota towards a recreational sector without addressing the underlying problems of limited regulation and data reporting could undermine the success that the QMS has experienced in protecting fish stocks.

The gradual allocation of fishing access to less regulated sports and recreational fishers who share the same fish stocks with quota holders, including Māori, weaken property rights and quota values. It contradicts the objectives of the Deed of Settlement 1992 and the spirit of collaborative management of marine resources. A variety of options can be considered to better incorporate recreational anglers into fishery management (Abbott, 2015). Key is a shift away from regulated open access that encourages increased fishing pressure and losses in the economic wellbeing of fisheries and ecological conservation goals.

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