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RESEARCH

Advancing ocean ecosystem conservation via property rights, rather than marine protected areas (MPAs)

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Abstract

There is demand to protect at-risk fish species and ecosystems. Property rights regimes can be superior to spatial controls via Marine Protected Areas (MPAs) for doing so. Empirical cases from Australia and the US indicate that MPAs are inequitable, too large and restrictive, and controversial. These conditions lead to resistance and political pushback, threatening long-term budgets and conservation goals. A critique of MPAs is presented along with a range of property rights arrangements–common, community, private—and Coasean bargaining as alternatives. Outlined benefits are a.) Rights holders have a stake in conservation and are central in its design. They are more than respondents. b). Costs/benefits can be more equally distributed, including direct payments that include both costs of transition and contribution to public goods provision. c.) Spatial set-asides confront tradeoffs and hence, are more apt to be economically sited and designed. d.) Modifications can occur more smoothly through market exchange than through the political process. Durable global conservation efforts can be enhanced.

Keywords Marine protected areas · Property rights · Ecosystems

Background: Marine Protected Areas (MPAs)

Over the past 25 years, Marine Protected Areas (MPAs) have been advanced as a means of safeguarding global ecological habitats and species at risk from excessive direct or indirect human use.[2] 14,688 MPAs currently exist (UNEP-WCMC and IUCN 2016), covering about 7.6% of global waters, about the size of North America, are within MPAs.[1] They vary in size, location, and nature, and range from less than 1km^2 to 1,500,000 km.[2] Some are pre-emptive, enclosing large, relatively pristine remote regions with no current exploitation, such as the 1,500,000 km^2 Papahānaumokuākea Marine National Monument northwest of Hawaii created in 2006 and expanded in 2016 by Presidential Executive Order and the 620,000 km^2 Kermadec Ocean Sanctuary northeast of New Zealand announced by the Prime Minister in 2015, but subsequently opposed by the Māori and currently stalled. Most MPAs, however, are in areas of existing human use and implement various levels of regulated access to address environmental degradation or dwindling fish stocks, such as the Apo Island and Sumilon Island Marine Reserves in the Philippines (Aliño et al. 2002). Other MPAs are more restrictive with no-take entry and exploitation controls, such as applied to parts of the Great Barrier Reef in Australia (Day 2016).

International efforts to establish or expand MPAs followed the Rio Earth Summit 1992; the 1992 UN Conference on Environment and Development (UNCED); the Convention on Biological Diversity 1993, ratified by 150 countries; and the World Summit on Sustainable Development 2002. The United Nations Framework Convention on Climate Change urged member nations to designate new marine protected areas by 2009. The 2017 United Nations Ocean Conference called for multinational action to conserve marine resources.[3] The International Union for the Conservation of Nature (IUCN) defined MPAs as "a clearly defined geographical space, recognised, dedicated, and managed, through

[1] In 2016, members of the International Union for Conservation of Nature (IUCN) called for protecting at least 30% of the ocean by 2030 through a network of marine protected areas (MPAs) The *Our Ocean* conference in Malta, October 2017, outlined MPA target coverage of 10% of the world's ocean areas by 2020 with subsequent expansion to 30% (http://ourocean2017.org/; Wood et al. (2008).

[2] https://www.protectedplanet.net/marine. The Marine Conservation Institute provides an MPA atlas and data base, https://marine-conservation.org/mpatlas/.

[3] https://oceanconference.un.org/callforaction.

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Table 1 IUCN MPA Protection Categories

Category Ia – strict reserve with most human entry and activity prohibited. No access and no-take MPA. Fully protected
Category Ib – larger reserves with wilderness protection constraints. No access and no-take MPA. Fully protected
Category II – specific ecosystem protection area. No-take MPA with limited use, tourism, recreation. Fully protected
Category III –small specific natural features. Special purpose MPA. Fully protected
Category IV – active habit and species management. Restricted MPA. Fully protected
Category V – protected landscape or seascape with recreation. Restricted MPA. Highly protected
Category VI – protected area with sustainable use of natural resources. Multiple use MPA. Highly protected

Source: https://mpatlas.org/glossary/; Kenchington (2016, 32–33)

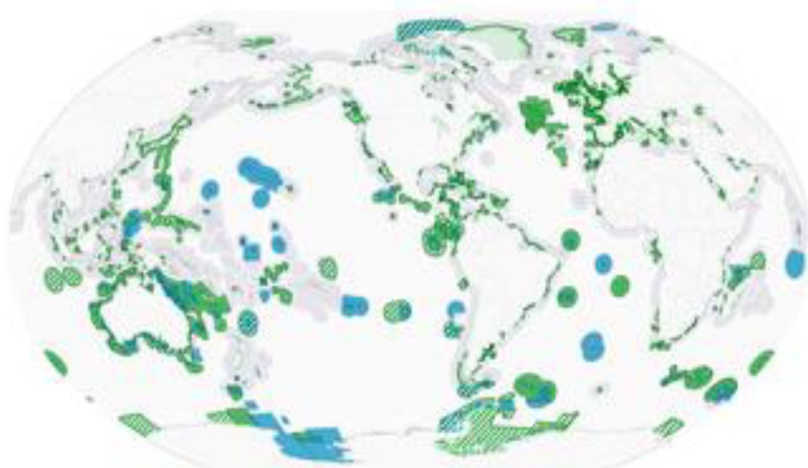

Fig. 1 Global Marine Protected Areas. *Source:* Wikipedia drawn from the Marine Conservation Institute, Marine Protection Atlas, https://en.wikipedia.org/wiki/Marine_protected_area

legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Spalding and Hale 2016 17).[4] The aim is to include 30% of the world's ocean areas by 2030, an area larger than Europe, Africa, and Asia combined. The US has approximately 1,000 MPAs, covering 26% of the country's waters, managed by NOAA (National Oceanic and Atmospheric Administration) with wide-ranging conservation objectives, including combating climate change. NOAA uses the IUCN definition of an MPA.[5]

The 2010 Parties to the UN Convention on Biological Diversity (CBD) adopted a Strategic Plan for Biodiversity 2011–2020 that included a ten-year framework to be implemented by all countries and stakeholders to achieve the Convention's 20 Aichi MPA Biodiversity Targets.[6] The IUCN provided a template for MPA design and corresponding zoning restrictions to be used in individual countries.[7] Aichi categories and restrictions on human activity are shown in Table 1.[8]

MPAs are depicted in Fig. 1 with highly restricted, no-take use areas shown in blue; those with limited entry and exploitation in green; and proposed MPAs in cross-hatched ocean regions. Virtually all MPAs are in country exclusive economic zones (EEZs) where respective governments have authority for designation and enforcement, and most are in waters adjacent to Australia, North America, and western Europe. Planned MPAs are to be in waters off Asia, Africa, South America, and elsewhere.

Welfare and conservation issues raised by MPAs

This section outlines the implications of directly imposing MPA regulations to achieve biological objectives. These actions can be viewed as Pigouvian controls (Pigou 1920). Although direct Pigouvian taxes are not applied,

4 https://www.un.org/depts/los/general_assembly/contributions_2014/CBD.pdf. https://www.st.nmfs.noaa.gov/ecosystems/ebfm/creating-an-ebfm-management-policy.

5 https://marineprotectedareas.noaa.gov/.

6 Aichi targets were adopted during the UN CBD summit in Nagoya, located in Japan's Aichi prefecture. https://www.cbd.int/undb/media/factsheets/undb-factsheet-sp-en.pdf.

7 See for example the MPA criteria used in Australia in Fitzsimmons and Wescott (2016).

8 See Cochrane (2016, Table 4.2, 55) and Day (2016, Table 5.3, 75).

environmental constraints are defined by governments and implemented by regulatory agencies as "polluter pays" restrictions. They are similar to taxes in that affected incumbent or potential users bear upfront costs, almost always without immediate, appropriate compensation. In some cases, fishery benefits are projected, but as described, these are uncertain with unclear time lines, and generally do not manifest in existing, well-managed fisheries. Further, they may not occur in a timely fashion, leading users to bear capitalized labor and capital costs in the interim. Compensating payments are rare and where they occur, are too small, relative to the public good claimed for the MPA.[9] Property rights and Coasean bargaining can address these issues.

MPAs in developed countries and likely in less developed ones, are mandated by legislation and regulatory policies to implement international accords. Actual national/state regulations are the outcome of the political process where relative lobbying strength determines outcomes.[10] They are not negotiated in a Coasean (1960) sense. Costs and benefits can be distributed unequally, violating CBD Aichi targets to achieve conservation in an equitable fashion. Political negotiations do not require *ex ante* and *ex post* economic cost/benefit analysis. Periodic programmatic reviews focus on biological objectives or in some cases, cost-effectiveness assessment. If using parties are made worse off, MPAs will not be welfare improving. Further, when incremental economic costs and benefits are not weighed in MPA design and implementation, spatial controls can be too extensive, large, and restrictive.

Political considerations

Although not all MPAs are consistent with Aichi 1a-ab, the most restrictive categories, they impose constraints on entry and use, and can be made more restraining. For this reason, incumbent users are wary of potential MPA designations in the ocean areas where their livelihoods depend. Accordingly, MPA advocates recognize the necessity of engagement with locals. This is not the same as granting them authority to block an MPA, but rather to inform how the process will play out with potential options for modification. It is reactive. Lubchenco et al (2003) for example state: "We define ''successful public process'' as one having scientific, political, and social integrity as well as durability over time." Integrity, however, involves a limited range of adjustments to meet economic concerns without compromising biological objectives.[11]

As with all government actions, there are distributional consequences. Politicians react to and depend upon the support of key constituents, and the regulatory bureaucracy requires the backing of political mentors and outside lobbyists for mandates and budget authorization, as well as information about species and ecosystem conditions (Karpoff 1987). Further, agency officials may be trained in the natural sciences, rather than in economics or other social sciences, and have a disciplinary tie to biological objectives. As tenured officials their livelihoods are little affected initially by MPA constraints, and they may have considerable regulatory discretion (Johnson and Libecap 1994, 154–171). Advancing either natural science or human economic concerns where they might compete in MPA designation, implementation, and management depends upon the relative lobby influence of the constituencies affected and the information provided to politicians and regulatory officials. Unless there are competitive interests, there are inherent biases in outcomes (Johnson and Libecap 2001).

In the empirical cases below, MPA proponents include members of environmental NGOs and international organizations, some affiliates of natural science associations and academics, as well as regulatory agency officials. Environmental advocates lobby government officials about MPA opportunities and obligations under international conventions, including the Convention for Biological Diversity and the International Union for the Conservation of Nature. Opponents include members of commercial and recreational fishing groups and inhabitants of their communities who stress potential negative impacts on commercial activities and their economic welfare.[12]

In lobbying politicians for MPAs, advocates may have an advantage. Although there is no empirical analysis of the political process behind MPAs, it seems plausible that members of environmental NGOs are more highly educated, have higher incomes, greater voter participation rates, and are more politically influential than are fishers and residents of fishing communities. The latter likely are poorer, have less

[9] In Australia there was compensation for transition costs to comply with MPA directives in some cases, as described below. There were no payments for public good outcomes.

[10] For example, consider ethanol which was once touted as providing triple bottom line environmental benefits. The legislative history is examined by Johnson and Libecap 2001.

[11] IUCN World Commission on Protected Areas (IUCN-WCPA). "Establishing marine protected area networks-making it happen." Washington, D.C: IUCNWCPA, National Oceanic and Atmospheric Administration and the Nature Conservancy; 2008:118. http://www.piscoweb.org/pub.

[12] For example, the Pew Charitable Trust launched the Global Ocean Legacy project in 2006 with the aim to establish the world's first generation of large-scale MPAs. See The Pew Charitable Trusts, "Archived Project: Global Ocean Legacy," http://www.pewtrusts.org/en/archived-projects/global-ocean-legacy. The key role of environmental NGOs, such as Pew, WWF, in lobbying for MPAs in Australia is underscored for example in discussions provided by Cochrane (2016, 46–50; Wescott 2016, 158–160).

education, and may be less active politically.[13] Further, if members of regulatory agencies are not disinterested parties, then MPAs have internal government support. Finally, in the political process, MPA proponents may have lower costs of collective action than do fishers. The former can mobilize around a single conservation objective, whereas fishers have far more heterogeneous goals and membership. They differ according to vessels and equipment, target species, location, and between commercial, sports, and recreational fishers.

When successful, proponents achieve their preferred conservation objectives via government action, while bearing few direct costs. Their costs involve organization and lobbying, but do not include changes in economic behavior or in livelihoods as is the case for those regulated by MPAs. As indicated in Fig. 1, MPAs often overlay areas of existing human activity with little or no compensation to offset regulatory controls. Absent reimbursement or timely, positive net benefits in fisheries or tourism, directly-affected parties are made worse off, and MPAs are unlikely to be aggregate welfare improving (Sallee 2019). There have been some adjustment payments for fishing groups in Australia, but none in the US example. Projected spill overs of biological stocks from MPAs to outside areas often are offered as compensatory benefits, but as noted, these may or may not play out in an opportune fashion.

There are clear problems of calculating exact public goods benefits of MPAs. Nevertheless, their potential can validate lobby efforts of proponents and any corresponding actions taken by politicians and agency officials. By contrast when distributional effects are not considered, MPA opponents can appear as obstructing the provision of global ecological goals for private gain. In evaluating conflicting claims of proponents and opponents, members of the broader public face high information costs, and have little incentive to search for actual MPA benefits and costs. Social science remedies for conflict between MPAs and users often call for greater interaction with other stakeholders to educate and create a common conservation view (Garces et al. 2013; Bennett and Dearden 2014; Voyer et al 2014, Cárcamo et al. 2014). Stakeholders, however, is an inclusive term, and the interests of various parties may not coincide with those of current ocean resource users.[14]

Spalding and Hale for example (2016, 16–17) describe engagement efforts with mixed results for MPAs in Australia. Outreach to affected parties also was called for in the Santa Barbara Channel Marine Park. As described in both cases, engagement was consultation that did not include the ability of users to fundamentally alter size, location, and Aichi constraints, including blocking the MPA altogether.

Imbalance in the distribution of costs and benefits

A mismatch in the distribution of MPA costs and benefits, as evident in the empirical cases examined below, leads to a lack of user compliance and support, and generates political pushback, undermining budgets and long-term conservation efforts. When resources are allocated via the political process with no tradable property right, then any revision and adjustment among competing stakeholders must occur via the political process with outcomes determined by relative influence. Outcomes are inherently uncertain, unstable and can be molded by perceptions of inequality.

One of Elinor Ostrom's (1990) key research findings as summarized by Cox et al (2010) for successful collective action in natural resource protection was the need for a proportionate distribution of costs and benefits among the parties involved. Disproportionate distributions encourage those, who receive more benefits than costs, to advocate more resource conservation than is cost-effective or welfare-improving in aggregate. By contrast, those who bare more costs than benefits, seek less action than would be appropriate and valuable in aggregate. Only balanced distributions encourage cohesion and advantageous collective action.

Similarly, Ronald Coase (1960) argued that automatic imposition of Pigouvian taxes and controls (Pigou 1920) to equate marginal social costs and benefits, placed all adjustment costs on the "polluter," and granted disproportionate benefits to the "pollutee." The resulting differential incentives lead "pollutees" to seek unwarranted outcomes, driving up costs, making marginal net social benefits *negative*, and lowering aggregate welfare. Moreover, such government policies do not provide easy remedy in the absence of authorized, low-cost exchange of regulatory instruments. As a result, Coase (1960, 18, 27) contended that an externally-imposed remedy for externalities could be more costly than the problem. Coase's counter was to acknowledge the reciprocal nature of externalities across polluters and pollutees, assign property rights, and allow for bargaining for mitigation. With exchange, marginal willingness-to-pay is

[13] See suggestive information in Smith (1994), Kahn (2002), Andrews et al (2010), Taylor (2014), and Coley and Mai (2022). Fishing communities are likely characterized by low income and education. In terms of the linkage between education, income and voting, representative is US Census Report (2021) on the 2020 US Presidential election.

[14] General citizens may bear indirect costs of possible increases in seafood prices or a rise in imports from MPA restrictions if they impact important fisheries, and by definition, secure only minimal portions of any biological public good. Broad citizen survey results showing support for MPAS are sometimes referenced in the literature, but general citizens do not bear direct costs. If they did, their

Footnote 14 (continued)

responses might be different. See discussion of "public" support, "engagement," and "multiple use" in Fitzsimons and Westcott (2016, 17, 91, 134, 174, 189).

equated with marginal willingness-to-accept, leading private marginal costs and benefits to be equalized, and serious imbalances in costs and benefits to be avoided.[15]

Costs are affected by MPA size; constraints, such as no-take with Aichi 1a; nature of infringement on existing fishing areas; spatial fish species densities across old and new areas; stock conditions at the time the MPA is designated; and location of alternative fishing locations and species.[16] They include lower harvests (catch per unit of effort) from fishing delays, prohibitions, and redirection; risk associated with new areas and species; costs of changes in fishing capital, labor, and markets; and costs from concentrated/competitive fishing along MPA boundaries. Congestion costs may also include damage to unprotected ecosystems if fishers rush to border areas and competitively exploit. This rush could undermine past, informal group fishery practices (Agardy et al 2011, 228–229). Further, there are learning costs associated with shifting to new areas; adopting unfamiliar new techniques, equipment, and labor; harvesting different species; as well as identifying new marketing outlets and shipping channels. These costs are capitalized over any adjustment period and are born directly by fishers and other using groups.

These user costs could be offset by increased subsequent productivity, such that better yields in new areas would compensate fishers for the losses incurred by closing certain fishing grounds. Offsets could occur if MPAs target spawning stock or nursery areas and create enough new production that spillover occurs. Spillover benefits depend upon MPA spatial boundary design, target species densities, existing stock conditions in and outside the MPA, recruitment, fish movements, as well as exogenous factors (Rasweiller et al 2012; Guenther et al 2015; Brander et al 2020). Based on their observations, Agardy et al (2011), however, are sceptical whether reserves can produce substantive spillovers fast enough to overcome both costs from physical displacement and perceptions of fishers that they are being unfairly restricted from traditional or the most productive fishing grounds.

In a literature review Kolding (2017) suggests that the evidence is limited of any increased yields from spillover and recruitment in a timely fashion for fishers. He concludes that MPAs are not optimal fishing management tools for sustaining fisheries or for replenishing fishing grounds with enhanced yields, as often is argued. For small-scale artisanal fisheries, where presently most new MPA emphasis is placed, the absence of benefits after bearing costs in initial lost yields when livelihoods depend upon fishing, would result in loss of support. This loss might be offset if tourism revenues rise, but these may be uncertain and not blend with historic skills and community social structures.

MPAs also require resources for implementation and maintenance across time. Those resources have opportunity costs that will be addressed politically in each country. Shifting budget demands in the absence of identified, measurable MPA net benefits and local political support weakens their position in budget allocation debates, and creates the potential for political reaction and risks for politicians and agency officials.[17]

In terms of benefits, MPAs are designed to protect at-risk ecosystems and species, but the public goods benefits are difficult to define precisely across accepted time periods for cost comparison. They also are subject to exogenous factors such as changing water salinity, temperatures, and shifting currents. It is possible that there may be uncontroversial ecological or species tipping points that the MPA is designed to avoid. Important benefit details, including biological triggers, timelines, measurable ecosystem and human gains, however, typically are simulated for prospective magnitudes and time.

Economic benefit assessment for cost comparison requires ecosystem valuation (Garces et al 2013; Rosales 2018). There are established methods in economics for valuing non-traded goods as indicated by Brander et al (2020). These techniques may or may not be used in initial MPA designation.[18] An alternative approach is to focus on cost measurement that can be more directly evaluated. Costs provide a benchmark for assessing ecosystem gains that would have to be at least be equivalent to costs for a benefit/

[15] Depending on market conditions, some trading parties may gain more surplus, so that their benefits relative to costs exceed those of other traders. Nevertheless, with voluntary markets and ease of entry and exit, these ratios would not be too different. On the other hand, with state regulation and distribution of costs and benefits, adjustments are political and far more costly. In another context, Hanich (2012) describes the problems of an imbalance in costs and benefits in fishery management.

[16] Suman et al (1999) identify costs associated with closed area zoning as part of the Florida Keys National Marine Sanctuary (FKNMS) that was implemented in 1997. Through surveys they found that the FKNMS had strong support from members of environmental groups. In contrast there were concerns from commercial fishers, who felt alienated from the process of zone designation and the costs of exclusion from historic fishing areas.

[17] In the MPAs examined here, there is little or no discussion of what adjustments would be made if anticipated biological linkages and outcomes do not appear. Would the MPA be dissolved or extended? What compensation would be provided to users who were restricted and bore costs, but benefits were not forthcoming? Alternatively, if no-take controls or other restrictions were very successful, would regulated-access and use be authorized if strict constraints were seemingly no longer required? If agencies do not have to bear opportunity costs, then such flexibility may not occur.

[18] This is a large literature. See Loomis and White (1996); OECD (2006); Dixon 2012 Grabowski et al (2012).

cost ratio to equal one and welfare improving (Ovando et al 2021).[19]

Absence of economic cost/benefit analysis

Ex ante and *ex post* economic cost/benefit analyses are not integral to MPA processes for two reasons. One is that they are difficult to do as indicated above with benefit and cost measurement. Second, given their legislative or mandated nature and overriding attention to biological objectives, there is little incentive among proponents to address economic calculus. Existing marine users are advisors, not actual decision holders, and cannot demand cost/benefit analysis as a condition of implementation. In none of the empirical cases examined below were economic trade-off studies undertaken as a condition for adoption or continuation. Pointing to three southeast Asia examples, Halpern et al (2013) describe a triple bottom line achieved in a conservation/equity trade-off. It is unclear, though, how equity is defined, which costs were incurred by which parties, or the time periods involved.

Periodic program evaluation is called for (Lester et al 2013; Ferraro and Hanauer 2014; Holland 2018). Program evaluation is not complete cost/benefit analysis if opportunity costs are not explicitly examined along with their distribution. Moreover, it is not the same as cost-effectiveness analysis that is sometimes noted (Halpern et al 2013). Cost-effectiveness analysis examines how predetermined conservation policy goals are achieved at least agency cost. Benefit/cost analysis in contrast, determines if or how an MPA would be implemented, adjusted, or abandoned relative to other options.

Davis et al (2019) discuss the challenges in estimating MPA costs and benefits, but do not provide empirical examples. Brander et al (2020) outline a framework for MPA cost/benefit analysis using updated values of ecosystem services for benefit measures, foregone fishery sales for costs, and a 3% discount rate. They do not provide analysis of the costs/benefits of specific MPAs, but rather suggestive findings for MPAs globally to meet CBD and IUCN targets. They attempt to account for spatial heterogeneity in ecological and economic conditions, and the findings are presented as generalizations.

Brander et al (2020) use value-transfer methods to evaluate ecosystem benefits broadly, and find that MPA benefits exceed costs by a factor of 1.4–2.7. They argue that targeting protection towards pristine areas with high biodiversity, yields higher net returns than focusing on areas with low biodiversity or areas that have experienced high human impact. While a reasonable conclusion, the aggregate nature of the approach and the very likely under-measurement of costs for displaced users, as well as limited benefits for well-managed fisheries, suggest that the conclusions may not apply to MPAs where there has been ongoing human use. Benefit measures also assume effective management and enforcement, which is dependent upon the distribution of costs and benefits.[20]

The absence of cost/benefit considerations creates a challenge for MPAs in achieving their conservation objectives and in insuring that they are broadly beneficial at the national level. Given the magnitudes involved in the 30% target, long-term country welfare considerations via trade-off analysis and political support are critical. While controversial in developed countries where most MPAs exist, they may be more so in developing, poorer countries, where fisheries and other resource users contribute importantly to local and national economies. Attractive fishery benefits may be better achieved by including ecosystem conservation in various rights-based fishery reforms as described below.

Empirical examples

Santa barbara channel case

Carla Guenther's (2010) study of the Channel Islands State Marine Reserve (CISMR) within the Santa Barbara Channel Islands Sanctuary illustrates the issues at hand with an unusual combination of biological and economic data. The CISMR is a network of 10 MPAs established in April 2003 within California State waters (0–5.6 km) around the northern Channel Islands, which are located 37 km offshore from the city of Santa Barbara. A marine reserve is defined in California law as an area of the sea in which consumptive or extractive uses are effectively prohibited and other human interference is minimized for ecosystem and species protection and diversity.[21] Adjacent federal waters were added in 2007. The authorizing California legislation included the Marine Life Management Act of 1998, the Marine Life Protection Act (MLPA) of 1999, and the California Ocean Protection Act of 2004 (Osmond et al 2010, 44, 49). The

[19] Ovando et al (2021) compare the costs of removing various levels of fish aggregating devices (FADs) in the central and western Pacific Ocean to reduce by catch of bigeye tuna by skipjack fishing vessels. Advocacy groups push for major reductions to achieve MSY, but estimated costs far outweigh likely benefits, suggesting reduced regulatory goals. A similar approach is used by Edwards et al. (2018).

[20] Bostedt et al (2020) provide a CBA study for temporary fishery closings, no-takes, in Sweden, but the positive results depend crucially on the ability of fishers to adjust smoothly and quickly. Visintin et al (2022) provide new estimates of ecosystem values in assessing a possible jump in benefits over costs for a MPA in Italy. They do not address costs in detail or whether the cost measures should be reevaluated.

[21] https://wildlife.ca.gov/Conservation/Marine/MPAs/Definitions.

regulatory agencies were the California Fish and Game Commission and the federal Fish and Wildlife Service and National Marine Fisheries Service in NOAA. 21% of Santa Barbara Channel Islands Sanctuary waters were placed in the CISMR as no-take.

Environmental NGOs were active in the reserves' legislation. A science advisory team was set up and a socioeconomic advisory committee was established. There was no economic cost/benefit analysis in initiation (Osmond et al 2010, 42, 43, 48). A socioeconomic impact analysis was conducted that assumed the total loss of all consumptive activity within marine reserves, but did not provide details of potential economic benefits resulting from conservation, such as tourism. Rebuilding fisheries was suggested, but was not the primary objective (Osmond et al 2010 49, 50).

The CISMR was implemented to reduce spiny lobster mortality, increase their harvest of sea urchins, and with lower urchin densities, protect kelp forests. There was no direct compensation for losses to fishers directed out of the MPAs. The affected spiny lobster fishery was one of the oldest commercial fisheries on the west coast with 60 active fishers, and had been well managed (Guenther 2010, 7). MPA biologists suggested that fishers would benefit from greater kelp densities and lobster stocks within the reserves and subsequent migration beyond CISMR boundaries where they would be available for harvest. Fishers, who would bear actual upfront costs were less enthusiastic in public hearings. They voiced concerns about the lack of scientific knowledge and consensus regarding reserve effects on fisheries which would impact any benefit predictions. They contested whether no-take reserves were effective for fisheries management and whether the predicted migration magnitudes and timing would occur.

Advocates based predictions on ecological population and community dynamic models that were influenced by variables are difficult to effectively model. These included lobster stock conditions within and outside the MPA, entry and congestion by fishers along boundaries, as well as natural, exogenous ecological factors affecting kelp forests beyond urchin grazing intensity. Timing was especially critical to fishers because of adjustments with ongoing capital and labor costs they would have to assume in response to MPA constraints. They could not block the MPAs nor seriously modify no-take restrictions once implemented.

Guenther (2010, 120) analysed surveys and catch panel data 5 years before and after the MPA designation in 2003. In terms of biological effects of no-take restrictions, she found that projected kelp cover and spiny lobster stock recoveries were less affected by the MPA and changes in lobster fishing pressure than by natural reef conditions and tide patterns. Further, she estimated that denial of access to past fishing grounds led to a 29% loss in individual daily catch associated with the direct loss of 17% of fishing grounds in the 5 years after MPA closures. There was no evidence of spill-overs from restricted grounds as an offset during the period she examined. The fishery impact was twice the magnitude of catch loss predicted by state and federal regulatory agencies when the MPA was under design in 2000 (Guenther (2010, 73). Lenihan et al (2021) find that over a longer period, spiny lobster stocks were stimulated by the MPA. This positive result, however, did not address the interim capital and labor costs faced by fishers.[22]

Following CISMR implantation, fishers engaged in costly search in less-known areas, some remote, requiring higher fuel costs and 20% more frequent, experimental, and expensive lobster pot baiting, setting, and pulling. Fishers also avoided previous fishing areas 1 km of MPA borders to avoid potential trespass penalties. 16 fishers left the fishery over the 10-year study with 14 exiting prior to MPA implementation, 4 departing as it took effect, and 2 left during the final 5 years. Interviews indicated that the MPA was a deciding factor in their departure (Guenther 2010, 121). Guenther concluded that those losses would have potential repercussions on harbor infrastructure, economy, and communities. Fisher households depended on lobster harvest for at least 50% of their income with 75% of fishers interviewed deriving 100% of household income from lobster fishing.

Guenther (2010, 33) concluded that her study had implications for the establishment of state legislated MPAs throughout California's coastal waters. She argued that MPA effects on kelp forests and biodiversity were variable and uncertain, while costs incurred by fishers were seemingly large.

Australia MPAs

The second empirical case examined is that of the Great Barrier Reef Marine Reserve (GBRMR) and other Australian MPAs. Australia's 272 MPAs cover around 36% of its ocean waters or 7,359,985 km^2 with about 10% of the MPA marine area in the IUCN's most restrictive protective categories, barring fishing, other entry, and exploitation (Kenchington 2016, 36).[23] Australia's MPA networks are shown in Fig. 2. The number of MPAs and proportion of state jurisdictional waters covered include 89 and 52% in

[22] Lenihan et al (2021) contend that MPA area reductions for the spiny lobster fishery of 35% after 6 years resulted in 225% increase in total catch, evidence that the restrictions benefitted the industry. Their study, however, does not fully account for trade-offs, particularly in the costs incurred by the fishery described in the text and the time within which migration occurred. Time involves capital and labor costs as noted above as well as other adjustment factors, born by fishers and the community.

[23] https://www.dcceew.gov.au/sites/default/files/documents/nrsmpa-protect.pdf. https://mpatlas.org/countries/AUS.

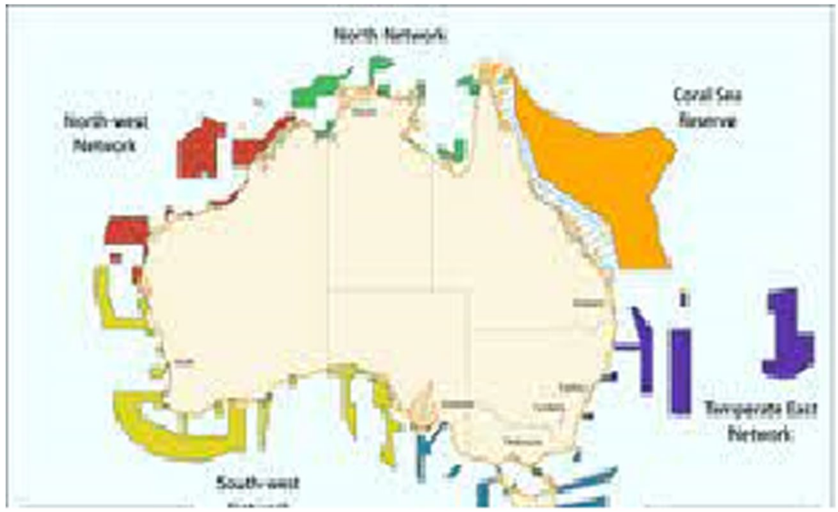

Fig. 2 Australia's Marine Protected Areas. *Source*: https://en.wikipedia.org/wiki/Australian_marine_parks

Queensland, 28 and 48% in South Australia, 18 and 40% in New South Wales, and 30 and 12% in Victoria (Kenchington 2016, 36, Table 3.2).

These empirical cases are instructive because Australia has the largest number of MPAs worldwide; has long experience with them, allowing for assessment; and cross-sectional evidence is available, absent with individual MPA case studies. The legal institutional setting is the same across MPAs. Consequently, general insights can be observed. Voyer et al (2014) claim that the Australian experience generally is representative of MPAs worldwide. They assert that absent local support, MPAs can fail or at least underperform when they are established primarily for biodiversity with fishery impacts considered as secondary.

MPAs and park reserves in all 6 states, 2 territories, and the federal commonwealth are included in a comprehensive survey in Fitzsimons and Wescott, eds (2016). The volume has 24 chapters, written primarily by ecologists in academics, government agencies, and environmental NGOs.[24] Additionally, one chapter is a commercial fishing industry assessment of MPA effects in southeastern Australia and the Great Barrier Reef (Boag 2016), and another addresses valuation of non-traded resources in MPAs (Hoisington 2016).

The authors assess MPA progress and complications as of 2015/2016. More recent assessments are consistent with those in the volume.[25] The MPA network began in 1975 with Great Barrier Reef Marine Park (GBRMP) followed by other MPAs, with most added prior to 2013. Overall, the conclusions are that although some biologically-sensitive areas are missing, the MPA process has stalled due to political reaction from user groups. For example, Clarke (2016, 184–187) claims that MPAs in New South Wales were added between 1997 and 2006, but halted with a moratorium on new areas between 2011 and 2015. Similarly, Ogilvie (2016, 211) points out that new or expanded MPAs in Queensland ended in 2011. In the Northern Territories Edyvane and Blanch (2016, Table 13.1, 219) state that most MPAs were declared prior to 2002 with some expansion in 2013. For Western Australia, Wilson (2016, Table 7.1, 124–128) describes early marine sanctuaries for humpback whale breeding and other ecologically sensitive species, established as early as 1987, but no further action after 2012.

Cochrane's overview of Australian commonwealth (not state or territory) MPAs (2016, 49–50) indicates that reserves were added to the MPA network based on biological objectives, but were opposed by commercial fishing and oil and gas interests that would be displaced. He concludes (2016, 56–61) that political opposition raised compliance and enforcement costs and threatened long-term adequate

[24] Environmental NGOs include the Australian Marine Conservation Society, Australian Wildlife Conservancy, Wildlife Preservation Society of Queensland, BirdLife International, World Wildlife Fund, International Union for the Conservation of Nature (IUCN), PEW, and The Nature Conservancy.

Academic institutions comprise University of Queensland, Charles Darwin University, ANU, James Cook University, University of Queensland, Deaken University, University of Wollongong, University of Tasmania, University of Technology Sydney, and the University of Western Australia. Government agencies represented are Western Australian Museum; Parks Victoria; Great Barrier Marine Park Authority; Commonwealth Marine Parks Review Commission; National Environment Science Program; Protected Areas agencies in Victoria, Queensland, and Northern Territory; Australia Antarctic Advisory Committee; Victoria Museum; Western Australia Marine Parks and Reserves Authority; South Australia Marine Parks Program; Queensland Park and Wildlife Service; Australian delegates to Convention on Biological Diversity; and MPA planning in Tasmania, South Australia, Northern Territory.

[25] https://www.uts.edu.au/news/social-justice-sustainability/australias-marine-unprotected-areas; https://theconversation.com/75-of-australias-marine-protected-areas-are-given-only-partial-protection-heres-why-thats-a-problem-149452.

funding, considering competing budget priorities, macroeconomic conditions, and shifting electoral cycles. In 2013 a new Australian government initiated a MPA review through 2016 and halted expansion of MPAs.

To understand this pattern of initial declaration, followed by halts or retrenchment, the authors point to distinctly different groups of proponents and opponents. The former included Australian commonwealth and state governments (state conservation councils), various academics, and members of Environmental NGOs, including WWF, PEW, and the Australian Marine Society. They moved aggressively to set up the MPA network to meet country commitments to the CBD (Cochrane 2016 50). The aim was to place as much area as possible into Aichi Highly Protected Categories, Ia-VI (Fitzsimons and Wescott 2016, 3–4; Kenchington Table 3.1, 31–33; Cochrane 2016 51, 55) with other areas in multiple use, and restricted human activities. They were motivated by CBD Aichi Target II of 2010 that sought by 2020 "…10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and *equitably* [emphasis added] managed…protected areas" (Kenchington 2016 30). Various intergovernmental agreements between the commonwealth (federal) and states/territories were held to devise a national strategy, including MPA targets, locations, and deadlines by 2012 (Cochran 2016 46–47).

The aims of the Convention on Biological Diversity (CBD) and Australia's political commitment to it "…to maintain healthy ecological function and manage the worst effects of human activity (Anderson and Laffoley 2016 vi) were challenged by users who bore immediate costs with limited or no compensation and uncertain forecast benefits. Cochrane (2016 49) notes: "Comprehensiveness of coverage from a perfectly scientific design perspective was *compromised* [emphasis added] by the accommodation of significant economic interests, notably commercial fishing…".

Proclamations of MPAs for biological purposes from 2004–2009 were followed by intense local reaction, despite the setting of local advisory groups and public discussion. Opposition is described by Wilson (2016, 134) for Western Australia and for South Australia by Thomas and Hughes (2016, 139–143). In South Australia with its designated 19 MPAs, the Marine Parks Council and Scientific Working Group called for no-take zones to cover 20–25% of each marine park. About 31% of South Australia's MPAs included IUCN's most restrictive areas (Thomas and Hughes 2016, Table 8.1, 145). The South Australia government attempted to reduce the impact on local commercial fishers with voluntary buyback of some fishing licenses and compulsory acquisition for others (Thomas and Hughes 2016, 147). Wescott (2016, 153–160) describes Victoria's 13 marine parks and 11 marine sanctuaries, designated as no-take that were set up in 2002. They were recommended by the Victorian Conservation Council and environmental NGOs to achieve the CBD Aichi Target II. Even though a limited compensation package was included, the MPAs became so controversial with opposition from commercial and recreational fishers that no further ones were designated.[26]

Boag (2016, 356–373), CEO of the Southeast Australia Trawl Fishing Association evaluates the declaration of commonwealth MPAs. He claims that the fisheries were well managed, not requiring MPAs. Boag (2016, 272–373) asserts that trawling in the area had little impact on the seabed, nor was there evidence of overfishing. Even so, between 2003–2015 between 39 and 44% of trawl fishing areas were closed. He argued that proponents did not understand the capital and equity costs facing fishers and that the value of fishing quotas and licenses were reduced by MPA restrictions, periodic closures, and overall uncertainty in access. A structural adjustment package by the commonwealth of $A220 million, including $A184 million in buybacks of licenses were insufficient. Boag (2016, 371) claims that there was no evidence of increased biomass after the MPAs were established.

Fitzsimons and Wescott (2016, 3) conclude that Australian MPAs have generated controversy and had a decline in political support. As remedy, Spalding and Hale (2016, 16–17) argue that more engagement is required for MPAs. If it includes the ability of local users to determine the adoption and nature of MPAs, including compensation, negotiated with advocates, then such engagement seemingly can move the process forward. Local buy in or acceptance of the feared misbalance in costs and benefits alone, however, might not resolve the distributional conflict.

Similar concerns have been documented elsewhere in Thailand (Bennett and Dearden 2014) and described by Mascia et al (2010), and Charles et al (2016). Where small MPAs are locally led to achieve ecosystem improvements with clear timely resident benefits, equity issues may not arise. Spalding and Hale (2016, 16) state that these arrangements are termed locally managed

[26] Kriwoken (2016 165) outlines MPAs in Tasmania where small no take reserves were implemented in a very politicized process. Kriwoken (2016, 167-168, 173) claims that the socio-economic impacts were not sufficiently addressed in the effort to meet IUCN targets and that fisheries were well managed, did not need no-take restrictions, and were important economically for the state. Main opposition came from commercial and recreational fishers and proponents were environmental NGOs. While there was early political support, in 2014 a new government stopped additional MPAs and reduced budgets for existing ones. Comparable experiences are described by Clarke (2016, 187) for New South Wales and Ogilvie (2016, 204-212) for Queensland. In the Northern Territories, Edyvane and Blanch (2016, 219-234) explain little MPA authorization due to opposition from recreational fishers and wariness by indigenous groups wary of government-imposed and managed sanctuaries.

marine areas (LMMAs), but may not meet MPA definitions under the CBD.

Great barrier reef MPA: design, extension, and compensation

Australia's Great Barrier Reef Marine Park or MPA (GBRMP) was established in 1975. It covered 350,000 km^2 and imposed restrictions on trawling, seasons, harvest, and minimum catch size. The area was labelled a World Heritage Site in 1981. Osmond et al (2010, 43–44) claim that the 1975 legislation passed by the Australian Parliament, establishing the GBRMP Authority provided a clear mandate and "unprecedented power" to maintain biological diversity, protect marine habitats, and to restore depleted or threatened species. Bioregions were defined by a scientific committee with 20% or more of the reef to be no-take areas. The objectives and plan were endorsed by the Australian Minister for the Environment. In 2004, the Australian and Queensland governments expanded the percentage of the park closed to fishing as no-take from about 5% to 33%. At the same time, the state of Queensland designated an additional protected zone. In total, 117,000 km^2 were placed off-limits (Day 2016, 65–74, 81). Zoning restrictions followed the IUCN Aichi designations.

In initial GBRMP design, proponents assured that the resident fishing industry would bear minimal losses (Stokstad 2015). New restrictive zoning in 2004 was added in part in response to lobby pressure by the members of environmental NGOs, the IUCN, and World Heritage Committee. The latter threatened to list the GBRMPA as "in danger" (Day 2016, Table 5.2, 5.3, Fig, 5.4, 72–76, 74–81). While biodiversity was the overriding objective, the zoning expansion was forecast to bring positive fishery spill-over benefits, and government compensation expenditures were provided. This is one of the few cases for worldwide MPAs where significant financial offsets were delivered. As part of a Structural Adjustment Package, the funds were paid through the buyback of fishing licenses, direct community payments, and subsidies for a switch to tourism. Expenditures ultimately cost the Australian government \$250 million AUD (McCook et al 2010; Macintosh et al 2010). Fishers still lost the value of harvest at A\$58 million annually.

The compensation has been challenged. Fletcher et al (2015) argue that the payments were too small and that large-scale expansion of no-take closures within the Great Barrier Reef did not enhance fishery production. Boag (2016, 370) concurs, challenging MPA projections that the 2004 closing of more than 28% of the Great Barrier Reef Marine Park would be compensated by rebounds in landed catch and value beyond the no-take boundaries within 3 years. He pointed out, however, that 9 years later, such offsetting rebound had not occurred and catch and landed value were down 33%. He asserts that initial fishery stocks had not been overexploited or depleted prior to the MPA so that there were limited migratory spill-overs from the restricted area. Davis (2015), however, argues that the fishery payments were excessive.[27] Regardless, the amounts are dwarfed by the national and global public-good gains, generated in some measure by adjustments in fishing. In an *ex-post* assessment Deloitte Access Economics (2017) and the Great Barrier Reef Foundation reported that the Great Barrier Reef generated \$56 billion in economic, social, and iconic value.[28]

If fishers had had a grandfathered property right as a group or as individuals to the reef and bargained with MPA proponents for trawling and other fishery changes, their willingness-to-accept would have been closer to a portion of the estimated \$56 billion in benefits than the transitional adjustments provided by the Australian government.[29] Further, in such a Coasean bargaining context, if advocates had had to pay for each additional area to be placed in the MPA or changes in restrictiveness, their willingness to pay likely would have been reduced for various ecologically-marginal ocean set asides or controls, lowering fishery impacts. Overall MPA benefits may not have been much affected, but the MPA likely would have been more suitably designed from a welfare, cost, and political sustainability perspective. A "fairer" allocation of benefits and costs would have resulted.

Property rights to ocean resource conservation

Property rights in general

Property, property rights, and markets have long been examined rigorously both in theory and in empirical analysis across many settings and time periods (Hayek 1945; Demsetz 1966, 1967; Cheung 1970, Williamson 1985, 2009; Barzel 1989; Libecap 1989; Ostrom 1990, 2009; North et al 2009; Acemoglu and Robinson 2012). The institution is well known. It is a ubiquitous, uniquely human custom that

[27] See also, Gunn et al. (2010) and Coggan et al (2022) for willingness-to-pay estimates, but not on lines discussed above.

[28] Deloitte Access Economics (2017). The \$56 billion apparently is a present value. https://www.barrierreef.org/the-reef/the-value#:~:text=More%20than%20the%20jobs%20it,economic%2C%20social%20and%20iconic%20asset.

[29] Libecap (2009) examines private bargaining between the city of Los Angeles and water rights holders in Owens Valley, California and shows that there was considerable surplus that might have gone to water rights holders had they had stronger bargaining organization. Historians and others have been critical of the imbalance. Such criticism is less apparent with imbalances in public good provision in environmental regulation.

underlies all economic activity in shaping expectations about resource control, use, and exchange. It is based on a moral notion of civil society that includes acceptance of equity and norms of right and wrong in access, use, and avoiding theft and trespass (Merrill and Smith 2007; Wilson 2020, 15). Property rights and markets require institutional formation and precision, and their specificity depends, in turn, on cost/benefit assessments (Demsetz 1967, Libecap 1978, Merrill and Smith 2007, and Smith 2012).

Applications include markets for ecosystem services, land easements and trusts, water quality permit exchanges and wetland mitigation banks, conservation banking, tradable development rights, and cap-and-trade air emission permits (Anderson and Libecap 2014, 134–72). In terms of conservation in fisheries and related resources, assigning ownership to existing users, fundamentally changes incentives for exploitation and conservation (Arnason 2007). Coase (1960) hypothesized that two self-interested parties would bargain to a mutually advantageous, Pareto-optimal level of an externality regardless of initial unilateral property right entitlements.

A variety of property rights institutions are available, depending on the setting (Hanna et al 1996; Schlager and Ostrom 1992; Schlüter et al 2020). They range from private individual property; group (common) property; community (common) property; spatial (common) property like TURFs in fisheries; and government (common) property. By assigning ownership to ecosystems and species, users are central to conservation decisions to establish, expand, and manage spatial conservation arrangements.

Conservationists seeking to protect specific areas, can negotiate to gain agreement from users on area set asides, bans on certain types of trawling or harvest, as well as limits on the inadvertent capture of non-target species and juveniles. Conservation becomes a joint effort, not a Pigouvian-style tax on one party. It is more likely to be Pareto improving and to secure lasting political support. Coasean-style negotiations determine payments and contractual arrangements. When paid directly for their contributions to the provision of public goods, users are motivated to assist in planning, implementation, management, and importantly, adjustment. Both parties have a stake in the outcome.

Compensating users is feasible because ecological and endangered species are increasingly valuable. Through ownership and exchange, they become assets, rather than threats. Monetized environmental assets can elicit the cooperation of those who know most about the ocean region and must adjust behavior as part of proposed conservation. Moreover, and perhaps even more fundamental, a property rights/exchange regime forces advocates to confront opportunity costs. Through bargaining, economic costs and benefits are weighed by balancing marginal willingness to pay and marginal willingness to accept. Users receive incremental net benefits for each area reserved or regulated. Conservationists receive incremental net benefits for each area set aside with differential constraints.

Exchange takes place so long as conservationists perceive value exceeding what fishers demand as compensation and so long as fishers perceive gains from incremental adjustments in harvest location, techniques, and species types. Flexible ongoing reaction to new costs and benefit information is feasible, relative to MPAs. Contracts can include updates as additional data appear that suggest changes in spatial coverage and fishing practices. Both parties have an incentive to negotiate. This market process makes conservation inclusive of key parties, welfare improving, and it distributes costs and benefits more evenly.

Common property

Common property, as outlined by Ostrom (1990), can have lower transaction costs than private property in definition and enforcement, especially for unobserved and unbounded resources (Allen 2000) and fewer equity conflicts. With resources held in common, division, marking, and enforcement of separate parcels are not required. Cross-parcel externalities from production and trade may be lowered.[30] Fairness concerns may be reduced because ownership is to the group, rather than to individuals. Broad resource enforcement costs may decline.

There are trade-offs, however, because internal decision-making rules must be devised and these may or may not be equitable. They may be cumbersome in operation. Majority rules, super-majority rules, or unanimity rules are examples of collective decision arrangements, each with different assignments of authority within the group and with progressively higher costs of administration and allocation, including market participation. By contrast, private property rights as outlined by Demsetz (1966, 1967) typically have the lowest decision-making costs because an individual or small group of individuals, each holding a share, decide on resource use and allocation. Free-riding is less feasible (Buchanan and Tullock 1962; Olson 1971). Consequently, markets perform most effectively and deliver their advantages most completely with private property (Demsetz 1966).

For these reasons, resources held in common may not be transferred easily via the market or group production be disciplined by market signals. Market exchange may upset local hierarchies, cohesion, and group-decision structures.

[30] Libecap and Lueck (2011) examine methods of parceling land to reduce externalities and promote production and exchange. Where the rectangular survey that they examine for surface land was implemented, land values were increased by 24%.

Property and production methods may be less flexible, values lowered, and potentially wealth may be reduced. Free-riding may be more common, requiring internal surveillance and enforcement resources.

Even so, the settings described by Ostrom (1990), Schlager and Ostrom (1992), Cox et al (2010), and Schlüter et al (2020) may be appropriate for common property and its application for ecosystem protection. Typically, group membership is small, homogenous in cost and resource objectives, and entry is restricted. Even where group membership is larger, but members are similar in production technology, equipment, organization size, and income from resource use, common property can operate effectively. Private property might not be feasible due to large numbers, small-scales, portioning costs, and boundary enforcement. Long-standing community arrangements, equity, and practices may be better maintained. TURFS as fishery management, for example, could be directed in a straightforward manner to conservation,

TURFS (Territorial Use Rights in Fisheries): common property for conservation

TURFs (territorial use rights in fisheries) designate ocean regions for collective management. They internalize valuable spatial externalities. TURFs are used in commercial and artisanal fisheries where there often are many small, similar fishers. Afflerbach et al (2014) compile information on 27 TURF-reserves worldwide, suggesting that strong customary tenure systems result in distinct, beneficial qualities of governance, management, and enforcement. Moreover, they can provide spatial ecosystem protection if specific species for protection are added for group maintenance and supervision (Cancino et al 2007). When TURF members are owners, they can negotiate with members of government agencies and environmental NGOs for non-target stocks and ecosystem conservation and compensation (Holland 2018).

The literature on TURFs and cooperation is large, primarily focused on fishery management, but some addressing collateral conservation controls (Deacon 2012; Wilen, et al 2012; Gelcich, et al. 2012; Ovando, et al. 2013; Holland 2018). While generally positive, in a northern Mexico case, McCay (2017) is more cautious on their benefits, but she does not address alternatives. Using spatial bioeconomic models, Kaffine and Costello (2011) and Costello and Kaffine (2017) outline how unitized or group efforts, such as those in TURFs, lower the costs of defining, managing, and enforcing harvest limits and marine preservation.[31] They describe hypothetical market exchanges between conservation NGOs and TURF organizations, and provide illustrative examples of effective private, spatial conservation in New York, California, and Chile. Where MPAs are in place, but controversial, extension to complete a conservation network could occur at lower cost with less opposition via TURFs.

Christy (1999) and Holland (2018, 471–77) describe the advantages of TURFs in fishery management in settings where individual transferable quotas are less likely to be effective. There are implications for the use of TURFs in conservation as improvements over mandated MPAs. They delegate ecosystem protection to users who can select cost-effect approaches and benefit on net from implementing them (Arnason 2008; Helson et al 2010). Collective management for conservation lowers bycatch, discard, habitat impacts, and spatial conflicts between user groups. To be effective secure catch and management rights (or privileges) to the collective group are required.

If ocean regions are relatively homogeneous, the group can create spatial harvest rules and practices to internalize external impacts of fishing; it can provide a framework for sharing information on location of vulnerable species/systems; it can pool risk and facilitate exchange among members who might inadvertently cause ecosystem damage, leading to harvest restrictions and area closures. Holland (2018) provides empirical examples of New England groundfish cooperatives that seek to lower the impact of trawling; of cooperative practices within the Bering Sea and Aleutian Islands bottom trawl fishery to reduce juvenile halibut mortality and pollock by-catch; of actions within the Pacific ground fish trawl fishery to avoid area closures; of efforts by the New Zealand Challenger Scallop Enhancement Cooperative and Deep Water Group to close areas within the New Zealand EEZ to bottom trawling for benthic protection (Helson et al (2010).

Private property rights for conservation, building on ITQS

Where individual fishery rights, such as individual transferable quotas (ITQs), individual vessel quotas (IFQs), or individual fisher quotas (IFQs) exist, they may have advantages over group ownership rights. Decision-making costs are lower; individual incentives are more completely incorporated; and if transferable, market reallocation and incentives for economic efficiency are enhanced (Scott 1999).

[31] See Wiggins and Libecap (1985) on the nature and benefits of unitization in natural resource management.

Key features of incentive-based fishing rights systems have not been incorporated in MPA policies.[32] They could be valuable alternatives for species and ecosystem protection. It is ironic that MPAs have not adopted rights-based approaches, given their background in replacing directed limited licensing, quotas, and equipment regulation. Incentive programs were implemented to replace mandated gear and harvest controls, beginning in the 1980s, after fishery regulation often failed to effectively control the race to fish. They followed insights from Gordon (1954), Scott (1955), and Christy (1973). Arrangements included calculation of total annual allowable catches (TACs) and assignment of catch shares of ITQs within them as a user right to fish. Where most successful, these share systems have documented improvements in fish stocks and incomes (Scott 1999; Hannesson 2006; Arnason 2005, 2008, 2012; Costello et al 2008, Costello et al 2010, Essington et al 2012, Costello et al 2016).

For example, using data from 4,713 fisheries representing 78% of reported global catch, Costello et al (2016) argue that reforms such as catch share systems could dramatically improve overall fish abundance while increasing food security and profits. ITQs also have incorporated tradable ecosystem shares. These practices could be expanded for ecosystem management previously delegated to MPAs (Holland and Schnier 2006). In a literature review Branch (2009) found that the impact of individual transferable quotas on ecosystems depended upon institutional design. Where ecosystem impacts were included, ITQs demonstrated benefits (Wallace et al 2015; Holland 2018; Reimer and Haynie 2018).

Holland and Schnier (2006) propose a system of individual habitat quotas (IHQ) to achieve habitat conservation and species protection. Individual habitat quotas would be distributed to fishers with an aggregate quota set to maintain targeted habitat stocks. As they describe, the system could be flexible to achieve a desired level of ecosystem quality without dictating the spatial distribution of fishing effort or habitat. Adjustment could be made to expand or contract habitat protection without the political process associated with MPAs. Their modelling indicates that an IHQ program is more cost effective for the protection of sessile non-target species than a fixed MPA.

There are variety of ways to implement IHQs, but one would be to link them to existing fishery quotas. If the habitat quota is met for a particular fisher, then unused quota could be secured from another to continue fishing. If overall habitat quotas were exhausted in a year, then fishing would be halted. Total quotas could be adjusted according to new information about the status of protected stocks. Alternatively, IHQs could be purchased by conservation advocates to raise their value and to encourage private species protection by each fisher, who might then conserve, release, and trade excess habitat quota.

Total allowable harvests and tradable quotas have existed since 1997 in the British Columbia bottom trawl fishery (Wallace et al 2015). Non-target species, such as cold-water sponges and corals, were added in 2012 with identification of high-risk areas, measurable milestones, and on-board and dock monitoring of harvests. Along with shares of target fish stock harvests, fishers are assigned shares of incidental or bycatch of non-target species or ecological resources.

As described above, once ecological shares are used, a fisher's efforts for target species must stop unless additional shares can be secured via trade from others who have surplus. Ecological resources and non-target species become assets. Conservation is encouraged because excess shares have value for trade. Reimer and Haynie (2018) examine the effect of Alaska Steller Sea Lion protection within a similar share system. Holland (2018) describes the use of incentive-based systems to achieve biological objectives in the Bering Sea and Aleutian Pollock and Pacific whiting fisheries in Alaska. Risk pools of bycatch quota that are exchangeable among members have been created to reduce the hazard of inadvertent harvest and potential target-fishery closure.

The use of ITQs and other property rights for ecosystem protection has critics. Equity issues are a major concern. The distribution of catch shares, which typically occurs by grandfathering (Anderson et al 2011), is viewed as unfair to those who lack a history in the fishery. Those who do, can be perceived as receiving an unearned rent windfall, while new entrants must buy their way in. There also is the impact of market trade on concentration of vessels and production. As noted above, property rights of some type–ITQs, IFQs, IVQs, community rights, and TURFs, have been adopted in light of the general failure of standard limited entry regulation to protect stocks and incomes (Grafton et al 2000 for example). Many regulated fisheries are characterized by redundant capital and labor in fishing (too many boats chasing too few fish); excessive storage, processing, and vessel support (fuel, ice, equipment services); and low-valued outputs. These investments, at least in the short run, sustain many small, remote communities, historical methods, and products.

With property rights and market exchange, traditional fishing operations and communities may be disadvantaged.[33] Crews may become smaller and processing plants may be

[32] For example, see the following: "Marine management and sustainable fisheries management are critical elements of good oceans management, but are not the same as protected areas management, where the primary focus is conservation of nature." https://portals.iucn.org/library/node/48887.

[33] See for example, Wilson et al. (1994), Acheson et al (2015), Bromley (2016), McCay (2017), and Young et al. (2018).

idled. The young especially may leave the industry and migrate away. Vessels may be consolidated and updated. Ports may change. Fresh versus frozen or canned fish require different and fewer production operations. While some small fishing communities wither, other fishing communities of course, grow, and new jobs are created. Small-vessel owners as well as recreational and sports fishers also are wary that fishery property rights may primarily benefit larger-scale, more capital-intensive, and often remote, commercial fishers.

There is no easy solution to these issues of economic and resource transition.[34] Whether traditional practices and communities can survive over the longer term as economic and fishing conditions change, as is likely, is very uncertain. Major subsidies might be required and whether or not they are politically viable is unknown. Property rights regimes have been adopted in the US for example since 1992 in a variety of ways across its many fisheries and regions to address these concerns. Some actions have restricted ITQ trade and added uncertainty in duration and security. Analyses indicate that these have reduced ITQ values and their possible effectiveness (Rieser 1997; Criddle et al. 2013; Grainger and Costello 2014).

This broad debate is beyond the scope and intent of this paper. For purposes here, the issue is whether MPAs or property rights regimes are more equitable and which better protects local users. The empirical record presented seems clear that MPAs in the US and Australia, at least, have imposed generally uncompensated costs on fishers and their communities with unclear benefits.

Finally, voluntary vessel and fishing license buybacks also are a vehicle for ecosystem protection (Holland et al 2017). Purchasers weigh the costs of buybacks with anticipated biological gains and fishers weigh payments with lost fishing opportunities.[35] In 2006 and 2007 the Nature Conservancy and Environmental Defense Fund acquired central California trawlers and groundfish permits with some retired and others leased back with restrictions on fishing techniques and areas to safeguard sensitive area ecosystems and species (Squires 2010). Seven federal trawling permits for commercial groundfish and four vessels were purchased and then leased back to fishers who complied with depleted species protections (Deacon and Parker 2009; Gleason et al. 2013). The exchange better reflected a balancing of benefits and costs than would mandated conservation controls.[36] In 2022 WWF-Australia bought and retired a commercial gill-net fishing license to protect dugongs, turtles, and dolphins in a northern Great Barrier Reef area of 100,000 km^2.

Implementation

There are mechanisms for shifting from MPAs to a property rights regime. It seems likely that existing MPAs would not be abandoned because of agency, NGO, and academic ties. Affected users may have abandoned the area or at least reduced their presence. Costs and any welfare effects have been born. But planned MPAs with ongoing fishing could be shifted to a property rights regime. An initial fishery property rights institution, such as a TURF or ITQ would have to be in place for exchange to occur along the lines laid out by Coase (1960). Environmental NGOs and government agencies could then negotiate with vessel owners or fishing organizations for changes in fishing practices in critical ocean areas. Fishing organizations would designate bargaining parties, limit entry, monitor compliance, and distribute relevant costs and benefits of any agreement. These are costly institutional arrangements, but the rising values of ecosystem assets would be offsets. Conservationists and fishers would bargain over value generation arising from protection. Each would have a stake in the process. Study would be required for determination of at-risk areas, targets, timelines, and enforcement. Overall, this procedure is comparable to the negotiated use of land easements for terrestrial conservation (Farmer et al 2011).

Concluding remarks

Conservation of unique marine ecosystems and species is of growing concern worldwide and is emphasized by multinational treaties, international organizations as well as by national governments. MPAs are a principal tool in this effort. They are posed to include 30% of the world's ocean area by 2030. Despite their broad public-goods objectives, they are unlikely to meet this goal. They are politically contentious in developed countries where most MPAs exist and likely will face high enforcement costs in less developed countries where expansion must take place. They may not be welfare-improving for human populations or advance the marine environment for long-term conservation.

[34] Economic transformation necessarily is disruptive, and fisheries are no exception as described by Hannesson (2006).

[35] Holland (2007) examine industry funded vessel buybacks. In ecosystem-valuable freshwater and land, the Nature Conservancy and Environmental Defence Fund, for example, purchase or lease land and water rights and reserve the resource for conservation uses. https://www.nature.org/en-us/what-we-do/our-insights/perspectives/water-for-life/

[36] Vessel buybacks and other forms of direct compensation to fishers also pay for losses in setting conservation goals, lower the costs of achieving those goals, and require balancing of trade-offs (Holland, Gudmundsson, and Gates 1999, 100; Holland, Steiner, and Warlick 2017; Squires 2010). The benefits of buyback, however, unravel if re-entry is not deterred.

As conservation directives, MPAs pose disproportionate costs on users and grant disproportionate gains to advocates. The setting encourages excessively numerous, large, and restrictive MPAs with few incentives for support or compliance among regulated populations or their political backing over the long term. MPAs do not make environmental resources assets for local protection, investment, and advance. Resident benefits may be limited. Directly-affected parties have little stake in MPA outcomes, and generally cannot capture returns from ecosystem improvements beyond asserted local fish-stock enhancement. These may or may not occur or be timely, depending upon the state of fish stocks, migration patterns, as well as broader exogenous factors.

Relative to a property rights option, MPAs are inflexible, not subject to marginal adjustments in size or restrictions considering new information. If indications suggest more controls or area are required, they would be opposed as the Australia case above indicates. On the other hand, considering the political/bureaucratic process by which MPAs are designated and managed, major downsizing also seems unlikely.

The remedy is a property rights regime for a more equitable distribution of costs and benefits, trade-off consideration, incentive-compatibility, and welfare-advancing ecosystem conservation. Property rights avoid non-negotiable, difficult-to-enforce, and unpopular directives. They allow for bargaining over conservation objectives and benefit/cost parameters. Property rights and Coasean bargaining make ecosystems valuable assets; encourage effective design of protections; and allow for periodic adjustments. Implementation builds upon well-known property and market institutions. They have been applied in incentive-based fishery management for stock and income gains and expanded to include non-target species and ecosystems. They are used for terrestrial conservation. Ecological property rights can be grandfathered to existing users. A property rights, market approach generates more effective, negotiated and enduring conservation that is consistent with human welfare and equity than do top-down directed MPAs.

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