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USER RIGHTS FOR OCEAN ECOSYSTEM CONSERVATION

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I. Spatial Conservation Regulation: Marine Protected Areas (MPAs).

International efforts to establish and expand MPAs for conservation follow from the 1992 UN Conference on Environment and Development (UNCED) and the Convention on Biological Diversity 1993, ratified by 150 countries. The 2017 United Nations Ocean Conference called for multinational action to conserve marine resources.[1] The International Union for the Conservation of Nature (IUCN) defined MPAs as "a clearly defined geographical space, recognized, dedicated, and managed, through 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).[2] The aim is to protect 30% of the world's ocean areas by 2030 from direct or indirect human use, an

[1] https://oceanconference.un.org/callforaction.
[2] 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.

area larger than Europe, Africa, and Asia combined. The 2010 Parties to the UN Convention on Biological Diversity (CBD) adopted a Strategic Plan for Biodiversity 2011–2020 with a ten-year framework to be implemented by all countries to achieve the Convention's 20 Aichi MPA Biodiversity Targets.[3] The IUCN provided a template for MPA design and corresponding zoning restrictions to be used in individual countries.[4]

I argue that in many cases MPAs do not appear to be efficiently designed: They can be too large and restrictive, may be inequitable, and may not deliver long-term conservation benefits. Political reaction may occur, leading to budget reductions and loss of aggregate support.

The paper proceeds as follows: The background leading to the adoption and spread of spatial conservation controls as Marine Protected Areas is described. International conservation conventions; various constraints on human entry and use in ocean regions; and the roles of global and national environmental groups in the process are briefly described. Because MPAs are viewed here as Pigouvian mandates, rather than negotiated Coasean controls, the political process is examined. It is argued that costs and benefits are unequally assigned between MPA advocates and opponents. Sources of political backlash to the ostensible provision of public goods is explored. Empirical examples to illustrate MPA processes and outcomes are presented for the US and Australia.

The MPA discussion is followed by a layout of a user rights alternative. User rights are viewed as property rights, even though they are more limited than formal ownership (Hannesson 2006, 23-27). Proposed institutional arrangements vary from tradable individual user/property rights to various types of common property, including augmented TURFS (Territorial Use Rights in Fisheries). Advantages over MPAs are described, and implementation is discussed. Concluding remarks summarize key points.

II. Spatial Conservation Mandates via MPAs

[3] 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.

[4] See for example the MPA criteria used in Australia in Fitzsimmons and Wescott (2016). See Cochrane (2016, Table 4.2, 55) and Day (2016, Table 5.3, 75) for Aichi targets.

14,688 MPAs currently exist worldwide (UNEP-WCMC and IUCN 2016), covering 7.6% of global waters, about the size of North America.[5][6] They vary in size, location, and nature, and they range from less than 1km^2 to 1,500,000 km^2. [7] Some are very large, 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 are smaller and in areas of existing human use and implement various levels of regulated access to address environmental degradation or dwindling fish stocks. The US has approximately 1,000 MPAs, covering 26% of the country's waters, managed by NOAA (National Oceanic and Atmospheric Administration), following the IUCN definition of an MPA.[8]Australia has the most extensive MPA system in the world, and major US ones lie off the Santa Barbara coast as examined below.

The IUCN (Aichi) zoning categories for MPAs restrictions vary, with some like Ia and Ib prohibiting all human entry and exploitation, while others like Zone V allow for tourism and VI authorize multiple, regulated uses. 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 to meet the 30% objective.

III. The Economics and Politics of MPA Spatial Controls.

As outlined by the objectives of the Convention for Biodiversity and the International Union for the Conservation of Nature, MPAs generally are imposed, Pigouvian-style mandates designed to halt ecosystem degradation and to protect species at risk. They typically place uncompensated constraints on current or potential users as "polluter pays" to achieve biological objectives. Although direct Pigouvian taxes are not applied, restricted users bear upfront costs as effective taxes on production. In this regard, MPAs are much like other environmental regulation

[5] In 2016, members of the International Union for Conservation of Nature (IUCN) called for protecting at least
[6] % 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).
[7] https://www.protectedplanet.net/marine. The Marine Conservation Institute provides an MPA atlas and data base, https://marine-conservation.org/mpatlas/.
[8] https://marineprotectedareas.noaa.gov/.

criticized by Coase (1960) where regulation assigns costs to users to reduce economic activity and thereby to provide environmental benefits. Trade-off analysis is not integral in Pigouvian approaches. The U.S. Endangered Species Act is an example (P.L. 93-205). As with Pigou (1920) compensation also is not part of the process. It is rare with MPAs and where it occurs, it appears to be too small, relative to the public good claimed for the MPA.

As illustrated in the empirical examples below, MPAs in developed countries like the US and Australia are mandated by legislation and regulatory policies to implement international accords. Actual regulations are the outcome of the political process where relative lobbying strength determines outcomes.[9] They are not negotiated in a Coasean sense. Costs and benefits are distributed unequally, violating CBD Aichi targets to achieve conservation in an equitable fashion. Using parties can be made worse off. Political negotiations underlying MPAs do not lead to ex ante and ex post economic cost/benefit analysis. Periodic programmatic reviews focus on biological objectives or in some cases, cost-effectiveness assessment. Absent careful cost/benefit analysis in design, the degree of restrictiveness may be set at an inefficient level.

A. Public Choice Considerations.

Although not all MPAs are consistent with the most restrictive Aichi 1a-1b spatial zones, 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. MPA advocates recognize the necessity of engagement with locals, but the latter do not have authority to block an MPA, but rather to inform and call for modification. The process is reactive.

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 (Stigler 1971, 1975; Peltzman 1976; 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 by MPA constraints, and they may have considerable regulatory discretion (Johnson and Libecap 1994, 154-171). Advancing either natural science or human

[9] 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).

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, and regulatory agency officials. Environmental advocates lobby government officials about MPA opportunities and obligations under international conventions, such as the CBD and IUCN. 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.[10]

In lobbying politicians for MPAs, advocates may have an advantage. Although there is no public choice 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 education, and may be less active politically. 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 across commercial, recreational, and sports fishers.

As with other mandated environmental regulation, MPA proponents achieve their preferred conservation objectives via government action, while seemingly bearing few direct costs. Their costs involve organization and lobbying, but do not include changes in economic behavior or on livelihoods as is the case for those regulated by MPAs. MPAs often overlay areas of existing human activity with little or no compensation to offset regulatory controls. Absent remuneration, directly-affected parties would be made worse off, and MPAs may not be aggregate welfare improving using the framework outlined by Sallee (2019). In the empirical examples below, there have been some adjustment payments for fishing groups in Australia, but none in the US example.

[10] 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).

Projected spillovers of biological stocks from MPAs to outside areas often are offered as compensatory benefits. These may or may not play out in a timely, cost/benefit fashion, and are dependent upon both endogenous and exogeneous factors, generating considerable uncertainty.

A Kaldor (1939)/Hicks (1939) criterion for compensation is implicit in MPA policies, where there are few actual payments. A Kaldor/Hicks criterion asserts that if policy benefits far exceed costs, actual compensation to parties directly constrained is not required to achieve aggregate welfare improvements. The global/local public goods delivery is assumed to be so significantly large that offsetting payments could be made, but are not necessary as policy justification. If accurate, an implied benefit/cost ratio would be well above one. In this case, economic cost/benefit analysis would support MPAs. There are obvious distributional effects and disgruntled parties can act to undermine provision of the public good, a factor not considered with Kaldor/Hicks. With large net benefits, however, compensation could be made and would be an outcome of Coasean bargaining.

There are problems in calculating public-goods benefits. Nevertheless, their potential can validate lobby efforts of proponents and 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 benefits for private gain. In evaluating conflicting claims of proponents and opponents, members of the general public face high information costs and may have little incentive to search for actual MPA benefits and costs. Social science remedies for conflict between MPA advocates and users call for greater interaction among affected stakeholders to educate and create a common conservation view (Garces 2013; Bennett and Dearden 2014; Voyer et al 2014; Ca’rcamo et al 2014). Stakeholders, however, is an inclusive term, and the interests of the various parties may not coincide with those of current ocean resource users.[11]

B. MPAs and an Imbalance in the Distribution of Costs and Benefits.

[11] As noted in Hannesson (2006, 170), 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 responses might be different. See discussion of public support, engagement, and multiple use in Fitzsimons and Westcott (2016, 17, 91, 134, 174, 189).

A mismatch in the distribution of MPA costs and benefits between lobby advocates and administrators, relative to regulated users is evident in the empirical cases examined below. It leads to a lack of user compliance and support, and generates political pushback, undermining long-term conservation efforts. When resources are allocated via the political process with no tradable property right, 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.

As argued by Ostrom (1990) and Cox et al (2010), successful collective action in natural resource protection requires 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. 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 a Pigouvian “polluter pays” tax (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. As a result, Coase (1960, 18, 27, 39) 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 equated with marginal willingness-to-accept leading private marginal costs and benefits to be equalized, and serious imbalances in costs and benefits avoided.

Costs are affected by MPA size; constraints, such as no-take with Aichi 1a-1b; nature of infringement on existing fishing areas; spatial fish densities across old and new areas; stock conditions at the time the MPA is designated; and location of alternative fishing locations and species. 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 costs could be offset by increased subsequent productivity, such that better yields in new areas compensate fishers for the losses incurred by closing previous 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 skeptical 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 historic, traditional, or 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 nor 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, could 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 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 identifiable, 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.[12]

[12] In the MPAs examined here, there is little elaboration as to 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.

C. Economic Cost/Benefit Analysis.

Careful ex ante and ex post economic cost/benefit analyses generally are not part of the MPA processes for two reasons. One is that they are difficult to do because of the challenges in 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 for implementation. In none of the empirical cases examined below were economic trade-off studies undertaken as a requirement for adoption or continuation.

Periodic program evaluation is called for (Lester et al 2013, Ferraro, Hanauer 2014, Holland 2018), but it is not cost/benefit analysis if opportunity costs are not explicitly examined along with their distribution. It is not the same as cost-effectiveness analysis. 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, not suited to cost/benefit analysis in specific countries where negative results are possible.

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 compliance and the distribution of costs and benefits.

The general 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.

IV. MPA Cost/Benefit Distributions and Implications for Conservation.

A. The Santa Barbara Channel MPAs.

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. [13]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.[14] The CISMR was established in California waters in 2003 and in adjacent federal in 2007 after authorizing legislation was enacted (Osmond et al 2010, 44, 49). 21% of Santa Barbara Channel Islands Sanctuary waters were placed in the MPAs 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 (Osmond et al 2010, 42, 43, 48). 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 by lowering urchin densities, protect kelp forests. There was no direct compensation for losses to fishers forced 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

[13] Guenther (2010, 6) argued that her study had broad implications for MPAs and fisheries beyond the CISMR.
[14] https://wildlife.ca.gov/Conservation/Marine/MPAs/Definitions.

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 fisheries management tools 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 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 beyond urchin kelp grazing intensity. Timing was especially critical to fishers because of needed 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) analyzed 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 occurred with no evidence of spill-overs from restricted grounds as an offset. The 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) subsequently 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.[15]

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 within 1 km of MPA borders to avoid potential trespass

[15] 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 period 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.

penalties. 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-year period. Interviews indicated that the MPA was a deciding factor in their departure (Guenther 2010, 121). Guenther concluded that 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.

B. **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).[16] The number of MPAs and proportion of state jurisdictional waters covered include 89 and 52% in 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 authors assess MPA progress and complications as of 2015/2016. More recent assessments are consistent with those in the volume.[17] 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 biologically-sensitive areas are still missing, the MPA process has stalled due to political reaction from user groups. For example, Clarke (2016, 184-187) claims that MPAs in New

[16] https://www.dcceew.gov.au/sites/default/files/documents/nrsmpa-protect.pdf. https://mpatlas.org/countries/AUS.
[17] 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-whythats-a-problem-149452.

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) show 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 MPAs (2016, 49-50) states 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 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 in Fitzsimons and Wescott (2016) 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 left in multiple use, but with restricted human activities. 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).

Proclamations of MPAs for biological purposes from 2004-2009 were followed by intense local reaction, despite the establishment 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 and 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.[18]

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 argues 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. Further Boag (2016, 371) claims that there was no evidence of increased biomass after the MPAs were established.

C. The Great Barrier Reef MPA.

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

[18] 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 halted 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 relatively little MPA authorization in the region due to opposition from recreational fishers and wariness by indigenous groups of government-imposed and managed sanctuaries.

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 (Day 2016, Tables 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 spillover benefits, and government compensation expenditures were provided. This, along with the payments noted above, is one of the few cases worldwide for 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 $A250 million (McCook et al 2010; Macintosh et al 2010). Fishers still lost the value of harvest at $A58 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 spillovers from the restricted area. Davis (2011), however, argues that the fishery payments were excessive.[19] 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.[20]

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

[20] Deloitte Access Economics (2017). The $56 billion apparently is a present value.

If fishers had had grandfathered user rights as a group or 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. Further, in such a Coasean bargaining context, if advocates had had to pay for each additional area to be placed in the MPA, their willingness to pay likely would have been reduced for various ecologically-marginal ocean set asides, lowering fishery impacts. Overall MPA benefits may not have been much affected, but the MPA likely would have been more suitably designed from a cost and political-sustainability perspective. A fairer allocation of benefits and costs likely would have resulted.

V. User Rights for Ocean Resource Conservation.

Although MPAs may sometimes achieve conservation goals in a collaborative manner, other policy options may be able to deliver greater ecosystem protection at a lower cost. It is very unlikely that the 30% ocean area set-aside Aichi target will be met. The problems with MPAs are outlined above: Incomplete weighing of economic trade-offs in establishment and implementation; an apparent inequitable distribution of costs and benefits; a generally nonoptimal fishery management institution; and political controversy, threatening budgets across political cycles, local support, and compliance, undermining long-term durability. In the empirical cases examined above, actual users are consultants, not principals in spatial conservation. A property rights regime could address these issues and help promote ecosystem protection.

A variety of property rights institutions are available depending on the setting (Schlager and Ostrom 1997; Schlütter 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 negotiate to buy compliance from users, including 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 regulatory tax on one party. It is more likely to secure lasting political support. Coasean-style negotiations determine payments and contractual arrangements. When paid directly

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https://www.barrierreef.org/the-reef/thevalue#:~:text=More%20than%20the%20jobs%20it,economic%2C%20social%20and%20iconic%20asset.

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. The exchange takes place so long as conservationists perceive value exceeding what fishers demand as compensation and so long as fishers perceive monetary gains from incremental adjustments in location, techniques, and species types. Flexible ongoing adjustment 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.

A. 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 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, 134-172). In terms of conservation in fisheries and related resources, assigning ownership to existing users, generally by grandfathering (Anderson, Arnason, and Libecap 2011), 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.

B. Common Property: Group and Community.

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 (Buchanan and Tullock 1962, Olson 1965). Consequently, markets perform most effectively and deliver their advantages most completely with private property (Demsetz 1966). Private property rights, however, may involve high transaction costs in definition, allocation, and enforcement (Allen 2000). High measurement and partition costs for unobserved or unbounded resources, as well as disputed equity outcomes affect the costs and benefits of private property rights assignment and trade.

Common property, as outlined by Ostrom (1990), might address both issues. With resources held in common, division, marking, and enforcement of separate parcels is not required. Cross parcel externalities from production and trade may be lowered. Equity 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, and they may be cumbersome in operation. Majority rules, or super-majority rules, or unanimity rules are examples, each with different assignment of authority within the group and with progressively higher costs of administration and allocation, including market participation. Further, beyond decision-making costs, property held in common may not be transferred via the market or group production disciplined by market signals. Market exchange may upset local hierarchies, cohesion, and group decision structures. Group property and production methods may be less flexible, values lowered, and wealth potentially may be reduced.

Free riding within the group can undermine adherence to decision rules, production, and cohesion. Internal enforcement is required. Hannesson (2006, 83) describes some of the challenges faced by common-property regimes. These costs underscore the importance of group size and homogeneity for determining effectiveness of the institution.

Even so, where group membership is small, homogenous in cost and resource objectives, and entry is restricted, common property may be appropriate. Where group membership is larger, but members are similar in production technology, techniques, organization size, markets, and income from resource use, common property also may be effective. In this case private property might not be feasible due to large numbers, small-scales, portioning costs, and boundary enforcement. Long-standing community arrangements and practices may be better maintained. TURFS as fishery management, for example, could be directed in a straightforward manner to conservation as outlined below.

C. TURFS (Territorial Use Rights in Fisheries) for Conservation.

TURFs designate ocean regions for collective fishery 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. They can provide spatial ecosystem protection if specific species for protection are included for group 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 within them is large. It is focused primarily on fishery management, but some TURFs address collateral conservation (Deacon 2012; Wilen, et al 2012; Gelcich, et al 2012; Ovando, et al 2013; Holland 2018). 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.[21] They describe hypothetical market exchanges between conservation NGOs and TURF organizations, and provide illustrative examples of effective private, spatial

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

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.

Indeed. group property rights arrangements can be more effective in internalizing incentives for ecosystem conservation than 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). 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 with implications for their use in conservation. 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 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; and 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).

D. User Rights for Conservation: ITQS.

Where individual user rights, such as individual transferable quotas (ITQs), individual vessel quotas (IFQs), or individual quotas (IQs) can be established at lower cost, they have advantages over group rights. Decision-making costs can be reduced; individual incentives can be more completely incorporated; and if transferable, market reallocation and incentives for economic efficiency can be enhanced (Scott 1999). Key features of incentive-based rights systems are not central in MPA policies, but seemingly are critical for species and ecosystem protection.[22]

[22] 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.

It is somewhat ironic that MPAs have not incorporated rights-based systems. As described, they are directed government regulations that incentive programs were designed to replace. The overall failure of mandated gear and harvest controls to protect fish stocks led fishery rights systems to be implemented beginning in the 1980s. 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 (Shotton 1999; Scott 1999; Hannesson 2006; Arnason 2005, 2008, 2009, 2012; Costello et al 2008, Costello et al 2010, Essington et al 2012, Costello et al 2016).[23]

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 fishery abundance while increasing food security and profits. ITQs also have been applied to 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 cost effectively and to better incorporate the information held by fishers. Individual quotas of habitat impact 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 habitat quality without dictating the spatial distribution of fishing effort or habitat. Their modelling indicates that an IHQ program with a conservatively established habitat objective is more cost effective for the protection of sessile non-target species than a fixed MPA. It allows for adjustment to expand or contract protection. There are variety of ways to implement IHQs, but one would be to link them to existing fishery quotas and if the habitat quota is met for a particular fisher, then that person could secure unused quota from another. If overall habitat quotas were exhausted in a year, then fishing would be halted.

[23] There are criticisms of ITQs based on distributional concerns (Acheson et al 2015, Bromley 2016). That debate is not entered here. Key, however, is that MPAs are clearly inequitable, so the relevant comparison is between an ITQ approach and existing MPAs.

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 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.

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.

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.[24] In 2006 and 2007 the Nature Conservancy and Environmental Defense Fund acquired central California trawlers and groundfish permits with some subsequently 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 have mandated conservation controls.[25] In 2022 WWF-Australia bought and retired a commercial

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[24] 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/ourinsights/perspectives/water-for-life/

[25] 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.

gillnet fishing license to protect dugongs, turtles, and dolphins in a northern Great Barrier Reef area of 100,000 km^2.

E. Implementation

There are mechanisms for shifting from MPAs to a property regime. It seems likely that existing MPAs might not be abandoned because of path dependence based on agency, NGO, and academic ties, budgets, as well as sunk fisher adjustments. 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 exist for exchange to occur along the lines laid out by Coase (1960). Environmental NGOs and government agencies could 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).

In areas where there is no current fishing, the ocean region of interest could be mapped and auctioned by governments for ecological benefits, in a manner analogous to US offshore oil and gas leasing (Hendricks et al 1993, Mead 1994). Additionally, in other areas, ecological rights could be grandfathered to existing or adjacent commercial users. The value of such rights would be based upon potential biological gains. Environmental NGOs or governments could purchase or lease ecological property rights to achieve their conservation objectives. This market process provides a framework for species protection that is equitable and incentive compatible.

VI. 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. Despite their broad public-goods objectives, they are unlikely to meet their 30% ocean set-aside goal. They are politically contentious in developed countries like the US and Australia, where most MPAs exist and likely will face high enforcement costs in less developed countries, where expansion must take place. They may not be advance the marine environment for long-term conservation.

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 generally do not make environmental resources assets for local protection, investment, and advance. Resident benefits may be limited. Directly-affected parties may have little stake in MPA outcomes, and generally cannot capture returns from ecosystem improvements beyond asserted local fish-stock improvements. These may or may not occur or be timely, depending upon the state of fish stocks, migration patterns, as well as broader exogenous factors.

A user rights regime could result in a more equitable distribution of costs and benefits and advance ecosystem conservation beyond what MPAs likely can do. It encourages bargaining over conservation objectives and benefit/cost parameters. It is more likely to make ecosystems valuable assets; encourage effective design of protections; and allow for periodic adjustments. Implementation builds upon well-known property and market institutions. User rights 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 granted to existing users. Where there is no current exploitation, ocean areas can be assigned to adjacent users or auctioned, much like off-shore oil and gas leases.

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