ANNUAL REVIEWS

Annual Review of Resource Economics

Economics of Marine Protected Areas: Assessing the Literature for Marine Protected Area Network Expansions

Heidi J. Albers and Madison F. Ashworth

Department of Economics, University of Wyoming, Laramie, Wyoming, USA; email: jo.albers@uwyo.edu

Annu. Rev. Resour. Econ. 2022. 14:533–54

First published as a Review in Advance on February 9, 2022

The *Annual Review of Resource Economics* is online at resource.annualreviews.org

https://doi.org/10.1146/annurev-resource-111820-024652

JEL codes: O2, Q22, Q28, Q56, Q57

Keywords

reserves, 30X30, spatial, conservation, prioritization, no-take, systematic conservation planning

Abstract

Marine protected areas (MPAs) provide both conservation and economic benefits. Recent international conservation actors have called for a dramatic increase in the area of MPAs from almost 8% to 30% of marine area by 2030 in a policy called 30X30. Both the economics and conservation science literatures consider MPA decisions and MPA impact, although the economics literature focuses on fishery economic outcomes. This review uses an optimization framework for MPA decisions as a lens through which to evaluate the economics literature on MPAs in the context of the 30X30 expansion decisions. We argue for more economic analysis of MPA policy questions, including those around siting, design, restrictions, and management choices; impact evaluation; responses of people; low-income country settings and incomplete enforcement; spatial settings without metapopulations; and conservation rather than harvest objectives.

1. INTRODUCTION

Marine protected areas (MPAs) impose restrictions on activities within specific areas of marine settings. Categories of MPAs range from no-take reserves with no permitted extractive uses to multiple-use MPAs that allow some use. As with terrestrial protected areas (PAs), MPAs provide conservation outcomes by protecting the resources and ecosystem functions in the MPA. Unlike terrestrial PAs, MPAs are commonly used as fishery management tools within a marinescape. For both conservation and poverty alleviation reasons, the United Nations' Sustainable Development Goal 14 aims for larger MPA networks worldwide. The Convention on Biological Diversity calls for MPAs to increase from 7.7% to 30% of marine areas in MPAs by 2030, in a plan called 30X30.

Because area targets can be met in infinite ways, economic analysis of MPA network expansions can contribute to efficiency and enable larger conservation and economic outcomes per spending. MPA economic outcomes range from fishery incomes to net social benefits, including nonmarket values, while conservation outcomes include improvements in the number of species, species populations or biomass, resilience metrics, and ecosystem processes. Given the vast areas involved; the potential to protect critical ecological systems and biodiversity to thwart the current extinction crisis; the economic values of fishing, tourism, and other ecosystem services worldwide; and multitudes whose well-being corresponds to marine ecosystem health, economists should find ample reason to engage with the questions of how best to expand MPAs. Economic analysis of MPAs can inform the full suite of MPA decisions, including design such as size, location, network configuration, restrictions, tourism infrastructure, and management such as reducing threats through incentives, monitoring, and enforcement. Many frameworks could be useful in informing MPA decisions, such as systematic conservation planning, reserve site selection, software like MARXAN, and MPA policy impact analysis. These widely used frameworks typically do not incorporate economic analysis approaches, even when addressing economic outcomes such as fishery harvests. While economic analysis of MPAs focuses on fishery outcomes, economic perspectives on people's behavior in response to MPAs are particularly important because responses determine both conservation and economic outcomes. This review uses an MPA optimization framework as a lens through which to evaluate the MPA economics literature in the context of the 30X30 expansion decisions.

2. ECONOMICS OF MARINE PROTECTED AREA DECISIONS: THE LENS AND THE LITERATURE

Our framework for assessing the MPA economics literature follows an optimization problem structure: an objective, choice variables, marinescape and ecological system characteristics as constraints and equations of motion, and socioeconomic and institutional constraints. The choice variables include MPA location, size, configuration, restrictions, and management. The setting's bio-geo-physical characteristics include the spatial distribution of ecological attributes, the spatial behavior of the ecosystem such as fish dispersal, and system dynamics. People's decisions in response to the MPA and its management create additional equations of motion due to their impact on the ecosystem. Budget constraints and local institutions form additional constraints. Proceeding from the objective function through these choice variables and constraints, we use this decision framework to assess the economics literature's status and contribution to MPA policy.

2.1. Objective Function

Beyond area targets, MPAs seek to generate conservation benefits, economic benefits, or both. Economists consider MPAs to maximize net social benefits or, more commonly, to maximize economic returns (Davis et al. 2019b). Despite the importance of tourism revenues from MPAs, those values are rarely considered in designing MPAs. Conservation economists and scientists view the objective of MPAs as providing a suite of conservation outcomes, including stemming the loss of biodiversity and ecosystem services, maximizing the number of species, creating MPA representation of diverse systems, increasing fish stocks for harvested or nonharvested species, resilience to climate change, and coral reef health (e.g., Almany et al. 2009, Bates et al. 2019, Davies et al. 2017, Game et al. 2009, Magris et al. 2018, Wilson et al. 2020). Most MPA economic analyses consider both a fishery economic goal and a conservation goal.

Economists prefer an objective function that maximizes net social benefits from MPAs, including market and nonmarket values, but the lack of data and valuation estimates limits such analyses. Valuation of MPA benefits, like coral reef functions or tourism, are not directly linked to decisions about MPAs (Beharry-Borg & Scarpa 2010, Schuhmann et al. 2019, Spalding et al. 2017). Choice experiments identify a large willingness to pay for recreation and habitat protection (Davis et al. 2019a) and for tourism improvements (Paltriguera et al. 2018) through MPAs. Stated preference analysis estimates the value of expanding an MPA (Stefanski & Shimshack 2016). Still, relatively few such valuation analyses of specific species or ecosystem services are available to inform MPA decisions.

Many economic analyses view MPAs as fishery management tools and emphasize MPA impact on fish harvests and revenues as an economic objective, while also considering fish stocks as a conservation goal (Albers et al. 2020a, Cornejo-Donoso et al. 2017, Sanchirico & Wilen 2001, Smith & Wilen 2003). Reithe et al. (2014) examine the impact of MPAs on producer and consumer surplus in a rare welfare analysis. In positive analysis, Sanchirico & Wilen (2001) and Smith & Wilen (2003) consider the impact of a reserve on marinescape aggregate harvests and on marinescape stocks. In normative analysis, Sanchirico et al. (2006) ask when establishing a no-take reserve will maximize marinescape profits. Albers et al. (2020a, 2021) determine the optimal MPA to maximize income, comprising fishing income and onshore income, and to maximize marinescape stocks. In these frameworks, MPA benefits to marinescape fisheries through fish dispersal trade off against the lost fishing within the MPA.

Yet, conservation goals may not coincide with the harvested resource. Reimer & Haynie (2018) evaluate the impact of MPAs/fishing closures to conserve Stellar sea lions on net revenue in marinescape fisheries. Capitán et al. (2020) compare MPA decisions across several goals, including maximizing income, extracted resource stocks, and unextracted sea turtle populations. Combining modeling and empirics, Sultan (2021) considers the MPA impact on fish populations, biomass, and catch rates. Conservation science analyses find limited impact of MPAs on non-harvested species (Endo et al. 2019, Montero-Serra et al. 2019, Welch et al. 2018). MPA tourism can also reduce conservation outcomes (Lopes et al. 2017, Monti et al. 2018, Velando & Munilla 2011). Hastings et al. (2017) show that marine reserves in a multispecies fishery can reduce bycatch without reducing yields of the target species.

Whether for a conservation or an economic goal, the impact of the MPA is a function of what would have occurred without the MPA, just as tropical PAs are evaluated for their avoided deforestation (Andam et al. 2008). Davis et al. (2019b) emphasize that impact analysis and predicting outcomes in order to make decisions focus on the changes created by the MPA, while clarifying that changes must be driven by the MPA and not influenced by other MPAs (Ferraro et al. 2019). Albers et al. (2020a, 2021) choose MPAs that maximize the avoided stock loss by calculating the loss with and without MPA aggregate stock. Klein & Watters (2020) compare outcomes to a counterfactual scenario with no MPA. Predicting or assessing the impact of MPAs relative to the no-MPA setting, however, faces considerable challenges both in before/after comparisons and in choosing comparator or control sites (Davis et al. 2019b). Ferraro et al. (2019) find much of the

MPA impact literature to be largely uninterpretable because it fails to adequately address both excludability, with related issues of bias and identification, and interference, with issues around interactions between MPAs and surrounding areas.

2.2. Marine Protected Area Choices and Control Variables

MPA choices go beyond a simple per location yes-no decision on PA status. As in the reserve site selection literature, the decision to place any particular site in a PA is a function of that site's characteristics and the characteristics of all other sites in the PA network. MPA choices include the specific location, configuration, and size of the MPA; the accepted uses and restrictions on uses within the MPA; ecological and economic management decisions, including monitoring and enforcement; and temporal choices. All of these choices face information gaps in how they influence the creation of MPA benefits, particularly for conservation benefits beyond fish stock metrics.

2.2.1. Size. Conservation area targets, as in 30X30, have raised interest and skepticism in large MPAs. Advantages of large MPAs include allowing interactions between different ecosystems within one MPA, more holistic management, and economies of scale in management, although monitoring costs could increase with area (Wilhelm et al. 2014). In exploring MPA size for highly mobile species, Cornejo-Donoso et al. (2017) use detailed models of fish movement to determine how different movement assumptions drive the optimal MPA size to maximize fishery benefits, suggesting that large MPAs can be effective in stock conservation and increasing fishery outputs. Vandeperre et al.'s (2011) meta-analysis finds mixed impact on fishery catch rates as a function of no-take reserve area. In a setting with biological hot spots, Schnier (2005) finds that the optimal size of the MPA reflects productivity of the hot spot and other areas, but also heterogeneity of productivity. In MPA models that choose MPA size, Yamazaki et al. (2015) and Albers et al. (2020a) find that larger MPAs can lead to lower aggregate stocks due to larger no-take reserves that are not well enforced and suffer illegal harvest; the MPA size interacts with management actions. In their empirical work, Arias et al. (2015) find that compliance with MPAs is higher with smaller MPAs. The choice of MPA size contains trade-offs between an area's impact on management costs, threats to the system, and benefits based on fish movement and ecosystem services, with remaining questions about when those trade-offs lead to preferences for large or small MPAs.

2.2.2. Location. In models and settings with spatial heterogeneity in either the ecological system or economic returns (including distance costs) to fishing, the location of the MPA contributes to its outcomes. Within metapopulation models with fish dispersal among patches, the ecological heterogeneity can include source patches of dispersal and sinks (Crowder et al. 2000, Kroetz & Sanchirico 2015, Sanchirico & Wilen 2001). Sanchirico & Wilen (2001) and Sanchirico (2004) find that locating a reserve on a source in a sink-source system leads to higher aggregate biomass and can lead to higher aggregate harvests under some conditions, whereas locating the MPA on the sink decreases aggregate harvests. In settings with density dispersal, MPAs increase aggregate biomass and can increase aggregate harvest. Krueck et al. (2017) examine location and other choices across larval dispersal patterns. Sanchirico (2004) emphasizes that MPA siting requires a marinescape perspective that reflects the characteristics of each patch and their connections through dispersal rather than basing MPA location choices on single patch characteristics.

Similarly, the distribution of fishers and heterogeneity in their distance costs influence the MPA location (Albers et al. 2020a, 2021). The appropriate MPA location differs across management budget levels because enforcement combines with distance costs to deter illegal fishing within

MPAs, which determines the MPA outcomes. The MPA location is also sensitive to whether the ecological boundaries of the marinescape are open or closed to fish dispersal (Albers et al. 2021).

For settings in which subpopulations of species cross jurisdictional boundaries, the MPA siting decision is complicated by the interests of both jurisdictions and the public good nature of the MPA. Strategic interactions lead to suboptimal MPA locations (Ruijs & Janmaat 2007). Under some conditions and species characteristics, transboundary MPAs can resolve the overexploitation of boundary-crossing fish and lead to higher stocks and harvests for both countries (Costello & Molina 2021).

2.2.3. Network configuration. Defining appropriate networks of PAs requires considering the dispersal connectivity between MPAs, ecosystem and species representation across the network, and redundant conservation to mitigate local extinction risk. Although the conservation ecology literature finds varied results for the spacing and sizing of individual MPAs in a network, it emphasizes fish dispersal distances in establishing networks (e.g., Metcalfe et al. 2015, Puckett & Eggleston 2016). Guidance to practitioners suggests separating marine reserves according to the movement of adult, juvenile, and larval stages of fish to promote connectivity (Green et al. 2013). Uncertainty regarding fish dispersal distances may reduce the optimal distance between MPA sites (Halpern et al. 2006).

Configuration also contributes to outcomes through nondispersal and nonecological considerations. Metcalfe et al. (2015) emphasize the interactions between the configuration, number, and size of MPA sites in assessing trade-offs including socioeconomic outcomes. Sanchirico (2004) finds settings in which one reserve provides lower rents than two reserves. Albers et al. (2020a) find that the number and location of equal-sized MPAs in an MPA network change nonlinearly with increases in management budgets. Beyond these studies, little economic analysis considers the complex spatial decisions of fishers in response to networks of MPAs to inform network decisions.

2.2.4. Restrictions, IUCN category, and zoning. Although most economic analysis considers no-take reserves, different categories of MPAs—as defined by the International Union for Conservation of Nature (IUCN)—permit extractive activities within MPAs, forming another control variable that contributes to MPA net benefits. Costello & Ballantine (2015) report that 94% of MPAs are not no-take reserves and allow some fishing, which they argue limits MPA contributions to biodiversity conservation. Other ecological studies describe higher biomass in no-take MPAs than partially protected MPAs (Sala & Giakoumi 2018). In terrestrial PAs, economists find that multiple-use PAs provide more protection against deforestation than do more restrictive PAs and hypothesize that people's threats to PA resources differ across IUCN categories of PAs in the context of incomplete enforcement (Pfaff et al. 2014). Similarly, Arias et al. (2015) find higher MPA effectiveness through fisher compliance in MPAs that permit but regulate fishing. Economists have not explicitly explored the impact of restrictions or IUCN category on people's responses to MPAs, but these empirical findings raise the question of opportunities to use that category decision for both conservation and economic goals.

Zoning of MPAs creates areas with different access and use restrictions. In multiple-use MPAs, Vilas et al. (2020) find ecosystem metrics such as species biomass, size, and diversity, and the economic metric of total catch, to be lower in MPAs with lower levels of protected status in their zones. Considering zoning decisions, Grantham et al. (2013) find that focusing on the no-take zone without addressing local community resource use leads to negative consequences for communities but the analysis uses implicit frameworks to define community needs and access. Green et al.'s (2013) MPA network design guide for practitioners calls for the creation of large multiple-use MPAs that contain zones with different permitted uses, including no-take reserves, while differentiating zones to exploit synergies across zones. In terrestrial settings, economic modeling of spatial decisions by extractors informs the appropriate size and location of zones around core restricted zones, but such frameworks are not incorporated into software or decision frameworks for defining zones and zone use restrictions (Albers 1996, 2010; Robinson et al. 2011, 2013).

In addition to having restrictions on fishing by amount or location, multiple-use MPAs can restrict who has fishing rights and what fishing gear is permitted. Fishing gear restrictions might include carrying capacity–damaging activities like dynamite fishing or fish growth–impairing gear like fine mesh nets.

2.2.5. Management. In addition to ecological management such as removing invasive species, generating net benefits from MPAs requires management of people. People create value and economic returns through fishing and tourism activities, but people also pose threats to species and ecosystems within MPAs. Many parks are labeled paper parks because insufficient management allows illegal and destructive actions within parks, including MPAs. Management decisions to mitigate human threat within MPAs typically fall into two categories: monitoring and enforcement of restrictions and incentives through compensation with monitoring.

Most analysis of no-take zones assumes that the zones deter all harvest in the MPA, without consideration of the enforcement costs incurred to deter harvest. By ignoring enforcement costs, such frameworks cannot inform management decisions about MPAs and can lead to false predictions of MPA impact, which generates inappropriate siting decisions (Albers et al. 2020a, 2021; Hallwood 2004). Using a spatially implicit framework, Yamazaki et al. (2015) examine relationships between the fraction of a marinescape in a no-take zone, enforcement against illegal harvest, and dispersal across the marinescape and find that no-take zones that are too large result in both high levels of illegal harvest and low aggregate biomass. Considering enforcement costs for no-take zones and territorial use rights for fishing programs (TURFs), Davis et al. (2015) find that higher species abundances occur through enforcement and determine ranges of prices and costs in which enforcement increases revenues. Englander (2019) finds that exclusive economic zones (EEZs) deter unauthorized fishing despite the potentially prohibitive cost of enforcing those zone restrictions. Albers et al. (2020a, 2021) choose the optimal MPA location and enforcement level across enforcement budgets. They find that ignoring the costs of MPA enforcement leads to MPAs that are too large and located in the wrong locations—particularly in locations close to ports and with high levels of pre-MPA fishing—to achieve income or fish stock goals. In addition, these analyses find that enforcement levels that do not deter all illegal fishing can be optimal for income-maximizing MPAs and budget-constrained optimal for avoided stock loss maximizing MPAs. That finding implies a link between MPA decisions about permitted uses, such as the IUCN classification, and enforcement. Yet, MPA analyses that explicitly consider enforcement's impact on people's decisions and on MPA effectiveness remain limited (Arias et al. 2015, Corrales et al. 2020, Thiault et al. 2019). The benefits of enforcement go beyond impact on biomass and harvest; Hall et al. (2002) find that recreational visitors to MPAs have an estimated willingness to pay of US$6/family for policies that reduce illegal fishing.

Although much of the MPA network expansion will occur in lower-income countries, little of the economics literature addresses widely used, incentive-based mechanisms to limit illegal extraction in MPAs in those countries. Alternative income-generating projects for potential MPA poachers can induce lower levels of illegal fishing by making fishing relatively less attractive or by reducing time available for fishing (Albers et al. 2021). Programs that compensate local people for reduced fishing access through benefits-sharing or payments, however, may not induce reductions in illegal fishing if the programs are not well linked to community pressure or to monitoring with consequences (Robinson et al. 2014). Community-based MPAs invoke Ostrom-style concepts of overcoming open access overexploitation through community institutions with implicit or explicit enforcement of community rules. Comanagement can enhance fisheries and increase conservation benefits (Guidetti & Claudet 2010, Voorberg & Van der Veer 2020).

The economics and conservation policy literatures focus on fishing as the main threat to benefits that MPAs seek to generate. Threats to conservation benefits other than those related to fish stocks include ecosystem conversion, such as mangrove deforestation, pollution from aquaculture and tourism, and damaging fishing practices such as dynamite fishing, which all require management actions other than enforcing fishing restrictions. In addition, management actions can not only prevent damages but also create benefits, such as management decisions about tourism infrastructure. Management decisions might also reflect differences between artisanal, commercial, and recreational fishers, with the importance of recreational fishers often overlooked in MPA analysis (Arlinghaus et al. 2019).

2.2.6. Timing choices and considerations. Spatial fishery models used for MPA analysis typically examine the steady state without the transition from a depleted starting point (Albers et al. 2020a, 2021; Sanchirico & Wilen 2001; Sanchirico et al. 2010). In nonspatial fishery frameworks, the optimal policy to transition to the steady state is simple: apply a moratorium on fishing until reaching the optimal long run stock size (Clark 1990), but spatial settings are not simple. The spatial forest economics literature finds both cyclical steady states and long complex transitions to the steady state because discrete space interacts with the dynamics of growth in nontrivial ways (Robinson et al. 2008, Swallow & Wear 1993). Sanchirico et al. (2010) tackle the issue of how best to use fishery management of effort levels, including moratoria on fishing, to allow a depleted resource to recover in a metapopulation. Their results find that the single-location's most rapid approach path is not optimal for locations linked by dispersal in a metapopulation, although sets of sites can form that path. Although their study is not economic analysis, Brown et al. (2015b) find that using static models to consider marine reserve costs to fisheries provides inaccurate results as compared to dynamic models, especially given the dynamics of larval dispersal. In empirics, Ovando et al. (2016) find long periods after marine reserve formation during which conservation efforts are costly to fishers. Smith & Wilen (2004) empirically estimate both short- and long-run responses to marine reserve formation and find that fishing effort responds quickly to reserves, with more exploitation outside the reserve that may prolong the transition to the steady state. These and other analyses emphasize that the recovery path matters but do not explore how that transition influences the location or other decisions about a permanent MPA (Corrales et al. 2020, Kaplan et al. 2019, Russ & Alcala 2004).

Although most MPAs are established into perpetuity, fishery closures that act like temporary MPAs are a common management tool for both fishery outcomes and conservation of particular species in what conservation scientists label dynamic conservation (Reynolds et al. 2017). Conservation scientists investigate rotating MPAs and evidence of species protection from temporary closures (Cohen & Foale 2013, Game et al. 2009). Temporary MPAs are increasingly important policy tools to protect migratory species, such as closing lobster fisheries for migratory right whales. Smith et al. (2020) find that short-term closures cause highly variable impact across time due to the dynamics of target species distributions. Similar issues arise at the multiyear level for dynamic MPAs to respond to ecosystem and species location changes over time, sometimes due to climate change (e.g., Cashion et al. 2020). Costello & Polasky (2008) use interannual spatial models to identify years in which a reserve is optimal, based on ecosystem dynamics. Englander (2021) reveals the role of temporary closures to protect juvenile anchoveta in Peru in providing information to fishers that leads to increased total juvenile catch. Given the common use of temporary actions for fishery management, spatially dynamic MPAs could prove a useful and acceptable policy tool in addition to permanent, stationary MPAs. Regardless, economic analysis to inform the location-duration of temporary, and perhaps spatially moving, MPAs remains limited (Albers et al. 2022a,b).

3. BUDGET CONSTRAINTS

As compared to terrestrial PAs, establishing MPAs faces lower start-up costs due to the lack of private ownership of marine areas but potentially higher ongoing costs associated with enforcing access and use restrictions (Bohorquez et al. 2019). Waldron et al. (2020) predict that low- and middle-income countries (LMICs) will face 70–90% of the implementation costs of achieving 30% PAs by 2030, which signals the need for international funding sources. Despite the prominence of paper parks and tight budget constraints for ongoing management and enforcement in lower-income countries, but also in wealthier countries, economic analysis of MPAs rarely considers those costs in evaluation nor in siting decisions. In addition, little is known about the level of budgets required to make MPAs effective at generating economic or conservation gains. Gravestock et al. (2008) use regression analysis of surveys in 36 countries to determine what drives budgetary needs in MPAs, finding that those budgets are higher with large MPA sizes and higher MPA visitor numbers.

Conservation finance searches for sustainable funding mechanisms to enable MPAs to generate conservation and economic benefits, but the issues are daunting. Bos et al. (2015) find that funding is too small, is not from diverse sources, underutilizes existing financial mechanisms, and is not incorporated into conservation planning. Constrained optimization frameworks for MPAs demonstrate that making MPA network siting and sizing decisions without considering ongoing management budget constraints limits MPA effectiveness by choosing the wrong sites and sizes (Albers et al. 2020a, 2021). Millage et al. (2021) demonstrate that Conservation Finance Areas that incorporate fishing and lease payments can provide better outcomes than MPAs that have limited budgets. Recognizing the lack of MPA financing, Ison et al. (2018) use surveys to determine stakeholders' willingness to pay and willingness to contribute time to community management of MPAs as a bottom-up financing tool. Similarly, user fees, comanagement mechanisms, and capturing willingness to pay to fund MPAs appear to be feasible financing options in Taiwan and Jamaica (Chen et al. 2014, Reid-Grant & Bhat 2009). Waldron et al. (2017) identify connections between conservation spending and biodiversity loss and find that the dynamic path of conservation finance reflects human development pressure and conservation spending's effectiveness. Tourism generates gate receipts, although many countries centralize gate receipts rather than use them as management budgets. Nonetheless, tourism provides a central mechanism through which governments and local communities can capture global values for biodiversity conservation, yet tourism-related decisions rarely enter MPA decisions (Waldron et al. 2020). Overall, the full suite of conservation finance mechanisms—including grants, bonds, philanthropy, and various market-based instruments—have yet to provide sufficient sustainable financing for MPAs or to be adequately integrated into MPA planning (Bos et al. 2015).

4. POLICY, SOCIOECONOMIC, AND INSTITUTIONAL CONSTRAINTS AND SETTING

Characteristics of the socioeconomic, institutional, and policy setting act as constraints or initial conditions for MPA decisions. The management policy setting of the marinescape is of particular importance in designing MPAs because that setting determines responses of people to the MPA. For example, with rents dissipated in an open access marinescape prior to MPA implementation, Sanchirico & Wilen (2001) depict the importance of the relative size of dispersal from the MPA to create benefits to fishers outside of the MPA. Costello & Polasky (2008) and Albers et al. (2021)

find that marine reserves cannot correct all open access issues but can provide increased profits or incomes as a function of marinescape management. Albers et al. (2021) find that nonspatial management policies such as licenses, gas taxes, and wage programs can augment the effectiveness of MPAs. Sanchirico (2002) finds that limited-entry fisheries lead to different reserve siting decisions than settings with complete entry, especially with heterogeneity in the system. Herrera (2007) explores interactions between temporary MPAs and imperfect compliance quota systems. Davis et al. (2015) characterize relationships between MPAs and TURFs, where TURF management is a function of spatial property rights enforcement. Costello & Kaffine (2010) find that MPAs can improve or diminish economic and fish abundance, depending on the coordination across TURFs in the marinescape, including the emergence of private MPAs and strategic MPA decisions. In addition, because distance costs influence fishers' decisions, the distribution of fishers or fishing ports with respect to fishing sites also influences MPA choices (Albers et al. 2020a, Smith & Wilen 2004).

5. ECOLOGICAL SYSTEM CONSTRAINTS AND CHARACTERISTICS

Because fish and people move across MPA boundaries, most MPA analyses consider both the MPA and the surrounding marinescape. In fishery economics, the ecological system is often depicted as a marinescape of patches of fish subpopulations, with dispersal among subpopulations in a metapopulation (Sanchirico & Wilen 1999), potentially with some patches as hotspots (Schnier 2005). That said, there are settings in which the scale of a subpopulation is larger than the scale of MPAs, settings in which the policy scale does not permit distinguishing between subpopulations, and other settings in which a metapopulation does not apply. Because of the spatial nature of defining marine reserves, understanding the spatial organization of species to be protected is important, even if the metapopulation description may be irrelevant in many, and potentially most, marine settings (Grimm et al. 2003). Armstrong (2007) expands on the simple ecosystem descriptions in a spatial MPA economic analysis to demonstrate opportunities for improvements in MPA implementation. In addition, many marinescapes have metapopulations of many species, and multiple fisheries, that can be reflected in MPA frameworks; for example, Hastings et al.'s (2017) analysis finds that marine reserves can increase yields from strong stocks and persistence of weak stocks (Kellner et al. 2007, Walters et al. 2008).

The spatial movement of the fish resource and the connectivity of the system are characterized in many ways, including the density or source-sink dispersal of adult fish and diffusion or current-driven larval stage dispersal with sedentary adults (Costello & Kaffine 2010, Grüss et al. 2011). Whether through modeling or empirical analyses, results show that the spatial behavior of the resource plays a significant role in MPA impact and MPA design (Albers et al. 2021, Armstrong & Skonhoft 2006, Cornejo-Donoso et al. 2017, Puckett & Eggleston 2016). Species with more complex habitat needs, such as coral reef fish, migratory sea turtles, and whales, that move through different settings, including nursery sites and breeding sites, require different characterizations of their dispersal patterns through a connected marinescape (Mumby 2006, Albers et al. 2021). Still, most economic analysis does not explicitly address the dispersal matrix of marine species between sites or subpopulations. Bauer et al. (2010) and Albers et al. (2022a) demonstrate the importance of the dispersal matrix itself for decisions about conservation of terrestrial species.

As in most renewable resource economics problems, defining the growth of the resource, particularly in relation to management actions and harvest, forms a critical aspect of the dynamic analysis of MPAs. Most economic models include either a logistic growth function or a Beverton–Holt growth function for harvested resource stocks. Using those growth functions, most economic research on MPA decisions reflects the decision at the bioeconomic steady state (e.g., Albers et al. 2020a, 2021; Costello & Kaffine 2010; Hallwood 2004; Sanchirico 2004; Smith & Wilen 2003).

For conservation benefits unrelated to fish stock size, the benefit functions may behave quite differently over space and time than typical fish growth and dispersal functions, and the conservation outcome may face threats from sources other than fishers' harvests. For example, a conservation goal of maximizing the number of species in an MPA requires the use of ecological frameworks that depict interactions among species and how species diversity responds to policy, such as protecting reefs from tourists. Similarly, conservation goals to improve stocks of nonharvested fish require ecological system models integrated with economic policy frameworks or ecosystem-based management approaches to assess policies, including, but not limited to, fishing restrictions. Alternatively, conservation goals to maintain ecosystem function may require addressing deforestation of mangroves.

6. PEOPLE'S RESPONSE CONSTRAINTS AND CHARACTERISTICS

The economics literature on MPAs is clear on the importance of considering people's—typically fishers'—responses to MPAs in order to make decisions about MPAs or to assess MPA impact (Albers et al. 2020a, Bennett et al. 2020, Dépalle et al. 2020, Hannesson 1998, Reimer & Haynie 2018, Smith & Wilen 2003, Sultan 2020, Zhang & Smith 2011). In particular, MPA economic analyses emphasize the importance of the spatial response of fisher behavior to the spatial policy of MPAs (Albers et al. 2020a, 2021; Sanchirico & Wilen 2001; Smith & Wilen 2003). Although conservation scientists describe a response of fishers to MPAs, those descriptions—both modeling and empirical—often fail to consider important aspects of people's decisions. In some cases, conservation scientists evaluate MPAs with assumptions of no fisher redistribution of effort (Selig & Bruno 2010, Sutcliffe et al. 2015) or redistribution of effort with total effort constant (Friedlander et al. 2017, Klein & Watters 2020, Savina et al. 2013). Smith & Wilen (2003) find that typical assumptions about fisher response bias predicted outcomes in favor of MPAs when fuller descriptions of fisher spatial behavior find other outcomes. For economists to inform MPA decisions, continued development of predictive models and empirical analyses that provide the microeconomics of fisher decisions is critical.

Many economic MPA analyses assume that people respond to MPAs by taking advantage of all arbitrage opportunities in open access settings (Sanchirico & Wilen 2001) and in well-managed marinescapes (Costello & Polasky 2008). However, these assumptions may not capture artisanal settings with low numbers of fishers (e.g., Madrigal-Ballestero et al. 2017). Economists typically assume fishers' location decisions are based on maximizing their expected returns across the set of possible locations, including travel cost, and trading off against other values such as onshore wages (Albers et al. 2020a, 2021; Smith et al. 2010). While some frameworks tacitly assume complete compliance with MPAs by dropping the MPA from fishers' possible locations, other frameworks include the expected fishing returns in the MPA as a function of monitoring and penalties (Albers et al. 2021). Frameworks with a fixed cost per site may interpret that cost as the distance cost to access each site (Sanchirico & Wilen 1999). Because each fishing vessel incurs the distance cost, however, these frameworks effectively model a representative vessel at each site, which is not equivalent to assessing the aggregate of multiple individual boat decisions with each boat covering that fixed cost. Albers et al.'s (2020a, 2021) articles discuss individual fisher behavior and the aggregate impact across the marinescape in response to MPAs and other policies. These frameworks, however, cannot accommodate a boat choosing several sites, which alters the distance-based fixed costs incurred and the location of PAs (Albers et al. 2020b). In addition, economic models do not typically address the dispersal matrix through which fish swim between subpopulations, but fishers worldwide respond to fish dispersing from MPAs by fishing the line at MPA borders, which alters persistence of species (Grüss et al. 2011).

Economic empirical analysis employs random utility models to estimate fisher location and harvest decisions. Smith et al. (2010) assess the opportunity costs to fishers imposed by a no-take reserve, in the short and long run, as a proxy for their willingness to support the marine reserve. Smith & Wilen (2004) estimate MPA responses, including home port choice, using a seemingly unrelated regression, which shows that switches to different ports reduce distance costs and reduce de facto reserves that are accessible from the new ports. Further, Zhang & Smith (2011) emphasize the importance of fishers' relocation of effort in response to marine reserves and define two critical issues for empirical analysis: heterogeneity in those responses and unobserved fish stocks. Smith et al. (2010) characterize the impact of marine reserves on heterogeneous fishers—including changes to their set of fishing locations, the abundance of target species, and the costs of fishing—and state that identifying the short- and long-run costs can inform the siting of marine reserves from a perspective of stakeholder support or opposition to the marine reserve. Using a panel data model, Smith et al. (2006) find that estimates of MPA impact on yields may overestimate benefits because they do not capture production functions and heterogeneity across vessels, space, and time. A homogeneous fleet assumption incorrectly estimates the reallocation of effort in response to spatial fishery closures and biases welfare estimates downward (Dépalle et al. 2020). Due to heterogeneity among fishers, fishing effort may not adjust spatially to equate marginal revenue rates among fishers and can lead to port changes (Holland 2000). Zhang & Smith (2011) find heterogeneity in fisher spatial effort substitution in response to two marine reserves, with increased fishing in surrounding areas. In contrast, Sultan (2021) uses a random parameter logit model to estimate fishing effort and finds that travel distance, water depth, and catch rate variation lead to lower levels of fishing effort near reserves, which explains higher fish stocks better than fish dispersal spillovers for their example in Mauritius. Beyond the reserve boundaries, Lynch (2006) describes heterogeneous distributions of fishing effort across space in response to habitat variability. This empirical work suggests that the simpler models of fishing effort allocation across space may not be sufficient for predicting the response to MPAs, the resulting outcomes, and therefore the choice of MPA sites and sizes.

In addition, many aspects of MPAs generate responses from both fishers and nonfishers that influence MPA outcomes. First, influencing fishing response, many MPAs use various benefits-sharing, alternative income-generating projects, and onshore wage opportunities to compensate local fishers for lost access to resources and/or to create incentives to induce fishers to reduce their effort in fishing (Albers et al. 2021). Smith et al. (2010) consider responses to MPAs with an onshore opportunity, as do other frameworks with an opportunity cost to fishing. Albers et al. (2021) consider how some such policies and programs influence fishers' decisions and how that changed behavior leads to different MPA choices and outcomes. Beyond projects, tourism is a primary market mechanism to influence local people's fishing decisions. In Costa Rica, fishers report that tourism wages and opportunities have provided incentives to reduce fishing effort (Madrigal-Ballestero et al. 2017). Pham (2020) discusses the MPA management decisions in lower-income countries with respect to using tourism to diversify local people's incomes while reducing marine resource activities. In contrast, anecdotal evidence suggests that employment in tourism near MPAs can attract nonlocal people, which limits the potential positive impact of MPA tourism on local people and on their fishing effort. Second, local and nonlocal people may undertake recreation activities, MPA employment, or tourism employment that does not alter their fishing decisions but generates value. Noneconomists' meta-analyses find that well-enforced, no-take and old MPAs typically provide human well-being, with economic and governance considerations for well-being, but negative impacts also occur in one-third of cases examined (Ban et al. 2017, 2019).

Mascia et al.'s (2010) literature assessment finds that MPAs contribute to food security, social well-being, and political power in local communities, with heterogeneous impact on different groups. Overall, considering the fishers' and people's responses to MPAs is an important aspect of designing MPAs to achieve conservation and economic outcomes.

7. SOLVING THE MODEL TO MAKE DECISIONS ABOUT EXPANDING MARINE PROTECTED AREAS AND NETWORKS

Although an optimization framework is used here as a lens on the literature, using such a framework forces the integration of ecological and economic decisions, including people's behavior, to achieve positive outcomes for conservation, social welfare, economic values, or their combinations. Even for pure conservation outcomes, MPA decisions must reflect the responses of people because people's reactions to the MPA influence conservation and other outcomes. People respond to all aspects of the MPA design decisions: The location, size, and configuration inform people's distance-based costs and the distribution of the resource across the marinescape; the type and level of restrictions within the MPA influence people's trade-offs among locations and activities; and management decisions, including enforcement to influence the net returns from actions within the MPA. Yet, in practice and often in the literature, MPA decisions are made somewhat sequentially, such as siting and sizing decisions followed by management decisions, and often without regard to budget constraints on upfront costs or ongoing management costs. In addition, much of the literature ignores management actions altogether and examines or predicts the outcomes from no-take reserves without considering the costs of, and behavioral responses to, enforcement. Sequential decisions and analysis of parts of the set of MPA decisions lead to unexpected outcomes, including less conservation than expected or higher costs per level of conservation (Albers et al. 2021). For example, siting an MPA prior to considering the costs of management leads to large and inappropriately configured MPAs in terms of producing the most conservation per unit of spending (Albers et al. 2020a). Although general frameworks such as systematic conservation planning approaches integrate many dimensions of conservation decisions on a marinescape, thinking about these decisions as a constrained optimization over space and time ensures that the integration enables trade-offs as a function of people's response to MPA decisions.

Overall, the MPA economics literature emphasizes the importance of the spatial behavior of both the resources in the marine ecosystem and people in the marinescape, given that MPAs are inherently spatial policies. In fish metapopulation settings, the economics MPA modeling literature predicts that MPAs are most likely to achieve conservation outcomes in settings where fishing effort outside of the marine reserve can be controlled and/or significant enforcement occurs (Albers et al. 2021, Costello & Polasky 2008, Hannesson 1998, Holland & Brazee 1996). These frameworks predict that MPAs are most likely to achieve fishery yield increase when sufficient heterogeneity between sites generates dispersal, such that the marine reserve acts as a source of fish to harvested areas, and when the MPA is implemented in significantly overfished marinescapes (Costello & Polasky 2008, Hannesson 1998, Holland & Brazee 1996, Pezzey et al. 2000, Sanchirico & Wilen 2001, Schnier 2005). In systems with heterogeneous distance costs and limited management budgets, optimal MPAs are small and occur at some distance from highly fished near-port locations to leverage distance costs and limited enforcement to deter fishing in the MPA and generate exit from fishing overall, often with incomplete enforcement of the MPA (Albers et al. 2020a, 2021). Empirical analysis of MPAs shows that simplified assumptions of fisher reaction to reserve placement may significantly overestimate both the conservation and fishery benefits of an MPA (Smith & Wilen 2003). Heterogeneity in the fleet type, boat gear, and spatial characteristics can lead to different fisher responses to reserve formation that can influence the transition to the steady state and the benefits of the reserve (Dépalle et al. 2020, Holland 2000, Smith & Wilen 2004, Smith et al. 2006, Zhang & Smith 2011). In summary, MPA decisions should plan for impact and effectiveness by reflecting both spatial ecological attributes and spatial decisions of people.

To take steps toward economic analysis that informs MPA network expansions, integrating econometric assessment of MPA impact with further analysis to describe fisher and nonfisher spatial responses to MPAs, particularly in heterogeneous groups and settings, will improve the economic efficiency of MPA decisions. MPAs produce benefits beyond those associated with changing fishing behavior that require more analysis. Given the proposed expansion of MPAs, randomizing early MPA network extensions to create an experimental setting for analysis would provide critical information about what characteristics of MPAs provide net conservation and economic benefits (Ferraro 2009, Ferraro & Pattanayak 2006). In addition to the challenges of convincing policy makers to embrace that sort of experimental design in MPA expansions, MPA impact analysis is complicated by the treatment of much of the marinescape through the spatial connections of the ecological and economic systems. Nonetheless, using rigorous methods to assess MPA impact is critical to generating efficient conservation policy (Ferraro & Hanauer 2015, Ferraro et al. 2019, Ferraro & Pattanayak 2006).

8. DISCUSSION

8.1. Where's the Economics of Marine Protected Areas?

In the last decade, economics journals have contained few articles with MPAs as the central topic, despite the emphasis on expanding MPAs in conservation policy. The reasons behind this paucity of articles could include editorial opinions that MPAs are not an important economic issue, that the relevant questions have been answered, and that the techniques used to explore MPAs are not appropriate for economics journals; or perhaps economists simply are not engaged with marine conservation issues. In contrast, the conservation science literature, including interdisciplinary journals, contains thousands of articles about MPAs, some with economists as authors. Although publishing in outlets with wide readership, particularly practitioners and interdisciplinary researchers, is important for furthering knowledge about MPAs and informing policy, the lack of articles in economics journals makes it difficult to evaluate which articles use economics-based methods in a rigorous and appropriate way because most such publications are not reviewed by economists. This issue is particularly important with MPAs because many conservation scientists undertake analysis about economic outcomes and MPA impact. Similar-sounding analyses may use different methods or assumptions, particularly about how fishers reallocate effort in response to MPAs, that then drive results.

In addition, economic policy impact analysis has grown markedly in recent years, with some of that work assessing PA effectiveness. For example, econometric park effectiveness analysis carefully considers the lack of randomness of PA boundaries; that people behave differently near borders, which fishing location decisions clearly demonstrate; and that boundaries produce edge effects and spillovers. Yet, dozens of conservation science articles describe conservation effectiveness and MPA impact analysis that do not consider these advances in impact analysis, such as comparing outcomes in unprotected areas (Friedlander et al. 2017, Selig & Bruno 2010) and comparing the outcome measures before/after an MPA has been placed without considering the counterfactual of no-MPA (Savina et al. 2013). Even within economic terrestrial PA analyses, few consider people's decisions rather than pixel characteristics, the heterogeneity of people and their actions, or the role of management actions like enforcement or incentives to deter PA use. In addition, the fishing location decision is different from terrestrial deforestation location decisions in that fishing in any site that accesses one subpopulation in a metapopulation has an impact on all sites that access that subpopulation. These and other characteristics of the use of MPAs can be modeled by economists and brought into impact analysis to establish standards for how fishers' economic decisions influence MPA impact analysis.

In addition to further economic analysis of MPAs in economics journals, some issues might be addressed by developing a set of MPA research and assessment guidelines from a blue ribbon panel that includes economists. Gaines et al. (2010) provide guidelines for both conservation and fisheries management, but a detailed characterization of the ecological processes is not matched by similar detail on the economic behavior of fishers and nonfishers or the socio-institutional setting. Overall, the most important aspect of MPA decisions that economists can bring to the current debate about expanding MPAs and MPA networks is a more in-depth understanding of the actions of individual people and how those actions aggregate across people, space, and time, including in settings with heterogeneous fishers and other marine users.

8.2. Low- and Middle-Income Country Considerations

Much of the expansion of MPA networks is likely to occur in the developing world, but little research uses developing country characteristics defined in the environment-development economics literature to examine those decisions (Marinesque et al. 2012). First, individuals make decisions within subsistence and semi-subsistence settings, missing or thin markets, household labor and capital constraints, budget constraints, and high poverty, which can contribute to distributions of effort. Second, many artisanal fishing communities manage resources as groups and work in cooperatives. Third, fisheries contain heterogeneous groups of fishers such as local small-scale fishers, local commercial fishers, and international fishers, with different gear and spatial decisions. Fourth, many LMICs' MPAs have small budgets for management, including enforcement, which raises issues of using siting and other decisions to improve MPA effectiveness in the face of illegal harvest and incomplete enforcement. Fifth, multiple-use MPAs and compensation mechanisms are common in LMICs to address concerns about income burdens on high-poverty communities. Lastly, many LMIC fisheries are low-data settings that limit options to estimate responses to MPAs.

No analyses of LMICs in the MPA economics literature address many of these considerations. Madrigal-Ballestero et al. (2017) survey artisanal fishers to develop insight into imperfect enforcement and the tourism labor market trade-offs facing fishers in Costa Rica. A series of articles explores fisher decisions in the presence of imperfect labor markets with wage opportunities, small numbers of fishers, incomplete enforcement, labor allocation decisions subject to a labor time constraint, and limited enforcement budgets for MPA decisions for community income goals, fish stock goals, and turtle conservation goals (Albers et al. 2020a, 2021; Capitán et al. 2020). Brown et al. (2001) explore a multicriteria decision framework in Tobago with stakeholder-driven weights on different criteria and emphasize the LMIC context through a focus on fishing, tourism jobs, and subsistence.

Analyses outside of the economics literature explore LMIC MPA decisions but in ways that may not adequately reflect people's decisions in such settings. As elsewhere in the systematic conservation planning literature, Kockel et al. (2020) use lost fishing grounds as the metric of impact on artisanal fishers rather than the fisher response to MPAs. To evaluate trade-offs between biodiversity and socioeconomic values, Teixeira et al. (2018) estimate fisher opportunity costs in data-poor regions but use habitat characteristics alone to estimate the spatial distribution of those costs. Grantham et al. (2013) compare several zoning and MPA configurations but use proximity between communities and restrictive zones as a metric to determine an equitable distribution of community impact.

As discussed above, moratoria or temporary MPAs are often used to allow stocks to recover, but such analyses tacitly assume that zero harvest is feasible and costless to achieve. In LMIC settings, a moratorium takes food and a primary income source away from fishing communities in which people have few alternative income-generating opportunities and little access to credit to enable consumption smoothing over the moratorium. Paying fishers for lost fishing income does not remove the fishers' incentives to fish, which means that the moratorium must be enforced, with costs weighed against the benefits of the fastest recovery of stocks. Sanchirico et al. (2010) find that allowing some fishing during the recovery period leads to slower recovery but with little negative impact, which could be evaluated for LMIC settings along with subsistence and market constraints and costs of enforcement during recovery. Brown et al. (2015a) consider short-term losses incurred as MPA resources recover, exploring combinations of size, timing, and species restricted to generate pathways that limit those losses, but they do not incorporate those issues into marine reserve choices.

Similarly, decisions about the IUCN category and MPA restrictions should also reflect the LMIC setting. No-take reserves may not be effective given limited enforcement budgets. In terrestrial settings, some multiple-use PAs provided higher conservation benefits than stricter category PAs that are not well enforced, which could be important in LMIC MPAs (Pfaff et al. 2014). As Tanzania's marine parks are required to generate biodiversity conservation and to alleviate poverty, some marine parks allow local people (insiders) to fish in the MPA, with limited enforcement against outsiders and in core zones, while others make payments to villages based on tourist gate receipts, which improves well-being but does not create incentives to comply with MPA restrictions (Albers et al. 2021).

Further economic analysis of what influences the spatial response of artisanal fishers to MPAs, what management actions and programs work to foster compliance with restrictions, and how various sets of restrictions inform fisher decisions, is necessary to facilitate the expansion of MPA networks across the developing world to generate conservation and economic benefits.

8.3. Conservation Economics: Beyond Harvest

Most MPAs have both conservation and fishery goals but few analyses treat these goals jointly, due, in part, to the lack of valuation for many outcomes. Analyses that design MPAs that are robust across various desirable outcomes could help achieve dual goals. A production possibility frontier analysis can assess the trade-offs between conservation and economic outcomes to provide policy guidance (Polasky et al. 2008). Where some nonmarket values are known, further trade-off analysis is informative, such as Sanchirico et al.'s (2013) use of willingness to pay estimates for Stellar sea lions to assess trade-offs between species conservation efforts and other benefits. Viewing MPA networks as portfolios allows for trade-offs within the MPA network, with areas generating conservation benefits, such as through well-enforced no-take reserves, while other areas generate economic returns, such as through multiple-use MPAs, with portfolio approaches also mitigating risk.

Because the primary goals of many MPAs include conserving species and ecosystems, conservation economists have much to contribute to MPA policy beyond the current focus on fishery impact (Shogren et al. 1999). Perhaps economists' biggest contribution to biodiversity and ecosystem conservation is the use of cost-effectiveness analysis, achieving a conservation goal in a least-cost fashion (Ando et al. 1998). Fishing opportunity costs of MPAs are a central cost to the MPAs, which means that some analyses have this cost-effectiveness characteristic, yet other costs and values also matter and are rarely incorporated in MPA economic analysis.

Conservation economists could consider conservation benefit functions beyond those based on fish stocks to improve MPA decisions for conservation outcomes. Both the spatial and dynamic characteristics of those conservation or ecosystem service production functions may look rather different from the fish growth and dispersal functions prevalent in the MPA literature, which implies that the MPA design—across all MPA choice variables—will also differ. In addition, the human threats that MPAs seek to limit include nonfishing actions, such as ecosystem conversion for aquaculture or reef damage from tourism. To make appropriate MPA design choices in settings where fish biomass is not the primary driver of conservation benefits and fish harvests are not the sole threat, economists could work in interdisciplinary groups to depict relevant production functions, define management cost functions, and assess the impact of MPA design and management choices on the responses of people who generate value, such as through tourism, and who generate conservation losses, such as through pollution. Bringing economic tools to bear on MPA choices to deliver conservation benefits can improve the efficient allocation of limited funds for that conservation and therefore provide more conservation benefits per budget.

9. CONCLUSION

MPAs are significant tools for the conservation of marine systems and, often, for fishery management. With recent calls to expand MPA networks to cover 30% of marine area worldwide, MPAs are increasingly important in terms of their size, their impact on economic returns from fishing and tourism, the number of people whose livelihoods and lifestyles interact with them, and their role in mitigating damage from climate change. In addition, MPAs represent the primary tool to address the current extinction crisis. Through their expertise in addressing people's decisions, economists have much to contribute to the extensive MPA conservation science and policy literature by conducting more modeling and empirical analyses of people's, not just fishers', responses to MPAs and the impact of MPAs on people's well-being and conservation outcomes.

The existing economics literature focuses on fishery income or harvests and provides limited analysis of alternative goals, such as species protection, or of costs other than fishing opportunity costs, such as enforcement, to inform the efficient use of conservation funding through planning and implementing MPAs. For economists to play a significant role in the proposed dramatic increase in MPAs worldwide, we define four priorities:

1. Examine low- and middle-income country MPA expansions with explicit recognition of the socioeconomic and institutional settings;
2. Develop fuller descriptions of the spatial response of fishers and nonfishers to MPAs, including in heterogeneous groups;
3. Develop and analyze frameworks that assess spatial ecosystem service production functions for MPA benefits, beyond fish stocks and harvests, as a function of people and MPA design choices; and
4. Undertake econometrically appropriate MPA impact evaluation and encourage the expansion of MPA networks in an experimental structure to identify what works.

These and other economic MPA analyses will enable MPA and MPA network expansions to provide higher conservation and economic outcomes during this exciting period of PA expansions worldwide.

DISCLOSURE STATEMENT

The authors are not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

ACKNOWLEDGMENTS

The authors are grateful to the Wyoming Excellence Chair funds for financial support for fieldwork and to the participants in the Environment for Development Initiative's Coastal Collaborative for insights, field visits, and funding.

LITERATURE CITED

Albers HJ. 1996. Modeling ecological constraints on tropical forest management: spatial interdependence, irreversibility, and uncertainty. *J. Environ. Econ. Manag.* 30(1):73–94

Albers HJ. 2010. Spatial modeling of extraction and enforcement in developing country protected areas. *Resour. Energy Econ.* 32(2):165–79

Albers HJ, Ashworth M, Capitán T, Madrigal-Ballestero R, Preonas L. 2021. MPAs and aspatial policies in artisanal fisheries. *Mar. Resour. Econ.* 36(4):341–67

Albers HJ, Kroetz K, Sims C, Finnoff D, Horan RD, et al. 2022a. Using the characteristics of seasonal migratory species to inform conservation policy analysis. *Rev. Environ. Econ. Policy.* In press

Albers HJ, Lee KD, Martínez-Salinas A, Middleton A, Murphy M, et al. 2022b. Ecological characteristics of seasonal migratory species to understand threats and conservation needs. *Rev. Environ. Econ. Policy.* In press

Albers HJ, Preonas L, Capitán T, Robinson EJZ, Madrigal-Ballestero R. 2020a. Optimal siting, sizing, and enforcement of marine protected areas. *Environ. Resour. Econ.* 77:229–69

Albers HJ, White B, Robinson EJZ, Sterner E. 2020b. Spatial protected area decisions to reduce carbon emissions from forest extraction. *Spat. Econ. Anal.* 15(3):280–98

Almany GR, Connolly SR, Heath DD, Hogan JD, Jones GP, et al. 2009. Connectivity, biodiversity conservation and the design of marine reserve networks for coral reefs. *Coral Reefs* 28(2):339–51

Andam KS, Ferraro PJ, Pfaff A, Sanchez-Azofeifa GA, Robalino JA. 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *PNAS* 105(42):16089–94

Ando A, Camm J, Polasky S, Solow A. 1998. Species distributions, land values, and efficient conservation. *Science* 279(5359):2126–28

Arias A, Cinner JE, Jones RE, Pressey RL. 2015. Levels and drivers of fishers' compliance with marine protected areas. *Ecol. Soc.* 20(4):19

Arlinghaus R, Abbott JK, Fenichel EP, Carpenter SR, Hunt LM, et al. 2019. Opinion: governing the recreational dimension of global fisheries. *PNAS* 116(12):5209–13

Armstrong CW. 2007. A note on the ecological-economic modeling of marine reserves in fisheries. *Ecol. Econ.* 62:242–50

Armstrong CW, Skonhoft A. 2006. Marine reserves: a bio-economic model with asymmetric density dependent migration. *Ecol. Econ.* 57(3):466–76

Ban NC, Davies TE, Aguilera SE, Brooks C, Cox M, et al. 2017. Social and ecological effectiveness of large marine protected areas. *Glob. Environ. Change* 43:82–91

Ban NC, Gurney GG, Marshall NA, Whitney CK, Mills M, et al. 2019. Well-being outcomes of marine protected areas. *Nat. Sustain.* 2(6):524–32

Bates AE, Cooke RSC, Duncan MI, Edgar GJ, Bruno JF, et al. 2019. Climate resilience in marine protected areas and the 'Protection Paradox.' *Biol. Conserv.* 236:305–14

Bauer DM, Swallow SK, Paton PWC. 2010. Cost-effective species conservation in exurban communities: a spatial analysis. *Resour. Energy Econ.* 32(2):180–202

Beharry-Borg N, Scarpa R. 2010. Valuing quality changes in Caribbean coastal waters for heterogeneous beach visitors. *Ecol. Econ.* 69(5):1124–39

Bennett NJ, Calò A, Di Franco A, Niccolini F, Marzo D, et al. 2020. Social equity and marine protected areas: perceptions of small-scale fishermen in the Mediterranean Sea. *Biol. Conserv.* 244:108531

Bohorquez JJ, Dvarskas A, Pikitch EK. 2019. Filling the data gap—a pressing need for advancing MPA sustainable finance. *Front. Mar. Sci.* 6:45

Bos M, Pressey RL, Stoeckl N. 2015. Marine conservation finance: the need for and scope of an emerging field. *Ocean Coast. Manag.* 114:116–28

Brown CJ, Abdullah S, Mumby PJ. 2015a. Minimizing the short-term impacts of marine reserves on fisheries while meeting long-term goals for recovery. *Conserv. Lett.* 8(3):180–89

Brown CJ, White C, Beger M, Grantham HS, Halpern BS, et al. 2015b. Fisheries and biodiversity benefits of using static versus dynamic models for designing marine reserve networks. *Ecosphere* 6(10):182

Brown K, Adger WN, Tompkins E, Bacon P, Shim D, Young K. 2001. Trade-off analysis for marine protected area management. *Ecol. Econ.* 37(3):417–34

Capitán T, Madrigal R, Albers HJ, White B. 2020. *Siting marine protected areas with area targets: protecting rural incomes, fish stocks, and turtles in Costa Rica*. Discuss. Pap. 20-08, Environ. Dev., Turrialba, Costa Rica. **https://repositorio.catie.ac.cr/handle/11554/10307**

Cashion T, Nguyen T, ten Brink T, Mook A, Palacios-Abrantes J, Roberts SM. 2020. Shifting seas, shifting boundaries: dynamic marine protected area designs for a changing climate. *PLOS ONE* 15(11):e0241771

Chen J-L, Chen J-Y, Chuang C-T, Lu H-J, Liu H-H, Lin Y-S. 2014. Developing a co-management financing mechanism to enhance the financial sustainability of marine protected areas in Taiwan. *Mar. Policy* 48:126–33

Clark CW. 1990. *Mathematical Bioeconomics: The Optimal Management of Renewable Resources*. New York: Wiley-Intersci. 2nd ed.

Cohen PJ, Foale SJ. 2013. Sustaining small-scale fisheries with periodically harvested marine reserves. *Mar. Policy* 37:278–87

Cornejo-Donoso J, Einarsson B, Birnir B, Gaines SD. 2017. Effects of fish movement assumptions on the design of a marine protected area to protect an overfished stock. *PLOS ONE* 12(10):e0186309

Corrales X, Vilas D, Piroddi C, Steenbeek J, Claudet J, et al. 2020. Multi-zone marine protected areas: assessment of ecosystem and fisheries benefits using multiple ecosystem models. *Ocean Coast. Manag.* 193:105232

Costello C, Kaffine DT. 2010. Marine protected areas in spatial property-rights fisheries. *Aust. J. Agric. Resour. Econ.* 54(3):321–41

Costello C, Molina R. 2021. Transboundary marine protected areas. *Resour. Energy Econ.* 65:101239

Costello C, Polasky S. 2008. Optimal harvesting of stochastic spatial resources. *J. Environ. Econ. Manag.* 56(1):1–18

Costello MJ, Ballantine B. 2015. Biodiversity conservation should focus on no-take Marine Reserves: 94% of Marine Protected Areas allow fishing. *Trends Ecol. Evol.* 30(9):507–9

Crowder LB, Lyman SJ, Figueira WF, Priddy J. 2000. Source-sink population dynamics and the problem of siting marine reserves. *Bull. Mar. Sci.* 66(3):799–820

Davies TE, Maxwell SM, Kaschner K, Garilao C, Ban NC. 2017. Large marine protected areas represent biodiversity now and under climate change. *Sci. Rep.* 7:9569

Davis K, Kragt M, Gelcich S, Schilizzi S, Pannell D. 2015. Accounting for enforcement costs in the spatial allocation of marine zones. *Conserv. Biol.* 29(1):226–37

Davis KJ, Burton M, Rogers A, Spencer-Cotton A, Pandit R. 2019a. Eliciting public values for management of complex marine systems: an integrated choice experiment. *Mar. Resour. Econ.* 34(1):1–21

Davis KJ, Vianna GMS, Meeuwig JJ, Meekan MG, Pannell DJ. 2019b. Estimating the economic benefits and costs of highly-protected marine protected areas. *Ecosphere* 10(10):e02879

Dépalle M, Thébaud O, Sanchirico JN. 2020. Accounting for fleet heterogeneity in estimating the impacts of large-scale fishery closures. *Mar. Resour. Econ.* 35(4):361–78

Endo CAK, Gherardi DFM, Pezzi LP, Lima LN. 2019. Low connectivity compromises the conservation of reef fishes by marine protected areas in the tropical South Atlantic. *Sci. Rep.* 9:8634

Englander G. 2019. Property rights and the protection of global marine resources. *Nat. Sustain.* 2:981–87

Englander G. 2021. *Information and spillovers from targeting policy in Peru's anchoveta fishery*. Work. Pap., Univ. Calif., Davis. **https://papers.ssrn.com/sol3/papers.cfm?abstract_id=3807560**

Ferraro PJ. 2009. Counterfactual thinking and impact evaluation in environmental policy. *New Dir. Eval.* 2009(122):75–84

Ferraro PJ, Hanauer MM. 2015. Through what mechanisms do protected areas affect environmental and social outcomes? *Philos. Trans. R. Soc. B* 370(1681):20140267

Ferraro PJ, Pattanayak SK. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLOS Biol.* 4(4):e105

Ferraro PJ, Sanchirico JN, Smith MD. 2019. Causal inference in coupled human and natural systems. *PNAS* 116(12):5311–18

Friedlander AM, Golbuu Y, Ballesteros E, Caselle JE, Gouezo M, et al. 2017. Size, age, and habitat determine effectiveness of Palau's Marine Protected Areas. *PLOS ONE* 12(3):e0174787

Gaines SD, Lester SE, Grorud-Colvert K, Costello C, Pollnac R. 2010. Evolving science of marine reserves: new developments and emerging research frontiers. *PNAS* 107(43):18251–55

Game ET, Bode M, McDonald-Madden E, Grantham HS, Possingham HP. 2009. Dynamic marine protected areas can improve the resilience of coral reef systems. *Ecol. Lett.* 12(12):1336–46

Grantham HS, Agostini VN, Wilson J, Mangubhai S, Hidayat N, et al. 2013. A comparison of zoning analyses to inform the planning of a marine protected area network in Raja Ampat, Indonesia. *Mar. Policy* 38:184–94

Gravestock P, Roberts C, Bailey A. 2008. The income requirements of marine protected areas. *Ocean Coast. Manag.* 51:272–83

Green A, White AT, Kilarski S, eds. 2013. *Designing Marine Protected Area Networks to Achieve Fisheries, Biodiversity, and Climate Change Objectives in Tropical Ecosystems: A Practitioner Guide*. Cebu City, Philipp.: US Coral Triangle Support Partn.

Grimm V, Reise K, Strasser M. 2003. Marine metapopulations: A useful concept? *Helgoland Mar. Res.* 56(4):222–28

Grüss A, Kaplan DM, Guénette S, Roberts CM, Botsford LW. 2011. Consequences of adult and juvenile movement for marine protected areas. *Biol. Conserv.* 144(2):692–702

Guidetti P, Claudet J. 2010. Comanagement practices enhance fisheries in marine protected areas. *Conserv. Biol.* 24(1):312–18

Hall DC, Hall JV, Murray SN. 2002. Contingent valuation of marine protected areas: Southern California rocky intertidal ecosystems. *Nat. Resour. Model.* 15:335–68

Hallwood P. 2004. Protected areas, optimal policing and optimal rent dissipation. *Mar. Resour. Econ.* 19(4):481–93

Halpern BS, Regan HM, Possingham HP, McCarthy MA. 2006. Accounting for uncertainty in marine reserve design. *Ecol Lett.* 9(1):2–11

Hannesson R. 1998. Marine reserves: What would they accomplish? *Mar. Resour. Econ.* 13(3):159–70

Hastings A, Gaines SD, Costello C. 2017. Marine reserves solve an important bycatch problem in fisheries. *PNAS* 114(34):8927–34

Herrera GE. 2007. Dynamic use of closures and imperfectly enforced quotas in a metapopulation. *Am. J. Agric. Econ.* 89(1):176–89

Holland DS. 2000. A bioeconomic model of marine sanctuaries on Georges Bank. *Can. J. Fish. Aquat. Sci.* 57(6):1307–19

Holland DS, Brazee RJ. 1996. Marine reserves for fisheries management. *Mar. Resour. Econ.* 11(3):157–71

Ison S, Hills J, Morris C, Stead SM. 2018. Sustainable financing of a national Marine Protected Area network in Fiji. *Ocean Coast. Manag.* 163:352–63

Kaplan KA, Yamane L, Botsford LW, Baskett ML, Hastings A, et al. 2019. Setting expected timelines of fished population recovery for the adaptive management of a marine protected area network. *Ecol. Appl.* 29(6):e01949

Kellner JB, Tetreault I, Gaines SD, Nisbet RM. 2007. Fishing the line near marine reserves in single and multispecies fisheries. *Ecol. Appl.* 17(4):1039–54

Klein ES, Watters GM. 2020. What's the catch? Profiling the benefits and costs associated with marine protected areas and displaced fishing in the Scotia Sea. *PLOS ONE* 15(8):e0237425

Kockel A, Ban NC, Costa M, Dearden P. 2020. Addressing distribution equity in spatial conservation prioritization for small-scale fisheries. *PLOS ONE* 15(5):e0233339

Kroetz K, Sanchirico JN. 2015. The bioeconomics of spatial-dynamic systems in natural resource management. *Annu. Rev. Resour. Econ.* 7:189–207

Krueck NC, Ahmadia GN, Green A, Jones GP, Possingham HP, et al. 2017. Incorporating larval dispersal into MPA design for both conservation and fisheries. *Ecol. Appl.* 27(3):925–41

Lopes PFM, Mendes L, Fonseca V, Villasante S. 2017. Tourism as a driver of conflicts and changes in fisheries value chains in Marine Protected Areas. *J. Environ. Manag.* 200:123–34

Lynch TP. 2006. Incorporation of recreational fishing effort into design of marine protected areas. *Conserv. Biol.* 20(5):1466–76
Madrigal-Ballestero R, Albers HJ, Capitán T, Salas A. 2017. Marine protected areas in Costa Rica: How do artisanal fishers respond? *Ambio* 46(7):787–96
Magris RA, Andrello M, Pressey RL, Mouillot D, Dalongeville A, et al. 2018. Biologically representative and well-connected marine reserves enhance biodiversity persistence in conservation planning. *Conserv. Lett.* 11(4):e12439
Marinesque S, Kaplan DM, Rodwell LD. 2012. Global implementation of marine protected areas: Is the developing world being left behind? *Mar. Policy* 36(3):727–37
Mascia MB, Claus CA, Naidoo R. 2010. Impacts of marine protected areas on fishing communities. *Conserv. Biol.* 24(5):1424–29
Metcalfe K, Vaughan G, Vaz S, Smith RJ. 2015. Spatial, socio-economic, and ecological implications of incorporating minimum size constraints in marine protected area network design. *Conserv. Biol.* 29(6):1615–25
Millage KD, Villaseñor-Derbez JC, Bradley D, Burgess MG, Lenihan HS, Costello C. 2021. Self-financed marine protected areas. *Environ. Res. Lett.* 16(12):125001
Montero-Serra I, Garrabou J, Doak DF, Ledoux J-B, Linares C. 2019. Marine protected areas enhance structural complexity but do not buffer the consequences of ocean warming for an overexploited precious coral. *J. Appl. Ecol.* 56(5):1063–74
Monti F, Duriez O, Dominici J-M, Sforzi A, Robert A, et al. 2018. The price of success: integrative long-term study reveals ecotourism impacts on a flagship species at a UNESCO site. *Anim. Conserv.* 21(6):448–58
Mumby PJ. 2006. Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves at seascape scales. *Biol. Conserv.* 128(2):215–22
Ovando D, Dougherty D, Wilson JR. 2016. Market and design solutions to the short-term economic impacts of marine reserves. *Fish Fish*. 17(4):939–54
Paltriguera L, Ferrini S, Luisetti T, Turner RK. 2018. An analysis and valuation of post-designation management aimed at maximising recreational benefits in coastal Marine Protected Areas. *Ecol. Econ.* 148:121–30
Pezzey JCV, Roberts CM, Urdal BT. 2000. A simple bioeconomic model of a marine reserve. *Ecol. Econ.* 33(1):77–91
Pfaff A, Robalino J, Lima E, Sandoval C, Herrera LD. 2014. Governance, location and avoided deforestation from protected areas: greater restrictions can have lower impact, due to differences in location. *World Dev*. 55:7–20
Pham TTT. 2020. Tourism in marine protected areas: Can it be considered as an alternative livelihood for local communities? *Mar. Policy* 115:103891
Polasky S, Nelson E, Camm J, Csuti B, Fackler P, et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biol. Conserv.* 141(6):1505–24
Puckett BJ, Eggleston DB. 2016. Metapopulation dynamics guide marine reserve design: importance of connectivity, demographics, and stock enhancement. *Ecosphere* 7(6):e01322
Reid-Grant K, Bhat MG. 2009. Financing marine protected areas in Jamaica: an exploratory study. *Mar. Policy* 33(1):128–36
Reimer MN, Haynie AC. 2018. Mechanisms matter for evaluating the economic impacts of marine reserves. *J. Environ. Econ. Manag.* 88:427–46
Reithe S, Armstrong CW, Flaaten O. 2014. Marine protected areas in a welfare-based perspective. *Mar. Policy* 49:29–36
Reynolds MD, Sullivan BL, Hallstein E, Matsumoto S, Kelling S, et al. 2017. Dynamic conservation for migratory species. *Sci. Adv.* 3(8):e1700707
Robinson EJZ, Albers HJ, Busby GM. 2013. The impact of buffer zone size and management on illegal extraction, park protection, and enforcement. *Ecol. Econ.* 92:96–103
Robinson EJZ, Albers HJ, Kirama SL. 2014. The role of incentives for sustainable implementation of marine protected areas: an example from Tanzania. *Int. J. Sustain. Soc.* 6(1–2):28–46
Robinson EJZ, Albers HJ, Williams JC. 2008. Spatial and temporal modeling of community non-timber forest extraction. *J. Environ. Econ. Manag.* 56(3):234–45
Robinson EJZ, Albers HJ, Williams JC. 2011. Sizing reserves within a landscape: the roles of villagers' reactions and the ecological-socioeconomic setting. *Land Econ*. 87(2):233–49

Ruijs A, Janmaat JA. 2007. Chasing the spillovers: locating protected areas in a trans-boundary fishery. *Land Econ*. 83(1):6–22
Russ GR, Alcala AC. 2004. Marine reserves: long-term protection is required for full recovery of predatory fish populations. *Oecologia* 138(4):622–27
Sala E, Giakoumi S. 2018. No-take marine reserves are the most effective protected areas in the ocean. *ICES J. Mar. Sci.* 75(3):1166–68
Sanchirico JN. 2002. *Reserve site selection in a limited-entry fishery*. Discuss. Pap. 02-69, Res. Future, Washington, DC. **https://media.rff.org/archive/files/sharepoint/WorkImages/Download/RFF-DP-02-69.pdf**
Sanchirico JN. 2004. Designing a cost-effective marine reserve network: a bioeconomic metapopulation analysis. *Mar. Resour. Econ.* 19(1):41–65
Sanchirico JN, Lew DK, Haynie AC, Kling DM, Layton DF. 2013. Conservation values in marine ecosystem-based management. *Mar. Policy* 38:523–30
Sanchirico JN, Malvadkar U, Hastings A, Wilen JE. 2006. When are no-take zones an economically optimal fishery management strategy? *Ecol. Appl.* 16(5):1643–59
Sanchirico JN, Wilen JE. 1999. Bioeconomics of spatial exploitation in a patchy environment. *J. Environ. Econ. Manag.* 37(2):129–50
Sanchirico JN, Wilen JE. 2001. A bioeconomic model of marine reserve creation. *J. Environ. Econ. Manag.* 42(3):257–76
Sanchirico JN, Wilen JE, Coleman C. 2010. Optimal rebuilding of a metapopulation. *Am. J. Agric. Econ.* 92(4):1087–1102
Savina M, Condie SA, Fulton EA. 2013. The role of pre-existing disturbances in the effect of marine reserves on coastal ecosystems: a modelling approach. *PLOS ONE* 8(4):e61207
Schnier KE. 2005. Biological “hot spots” and their effect on optimal bioeconomic marine reserve formation. *Ecol. Econ.* 52(4):453–68
Schuhmann PW, Skeete R, Waite R, Lorde T, Bangwayo-Skeete P, et al. 2019. Visitors’ willingness to pay marine conservation fees in Barbados. *Tourism Manag*. 71:315–26
Selig ER, Bruno JF. 2010. A global analysis of the effectiveness of marine protected areas in preventing coral loss. *PLOS ONE* 5(2):e9278
Shogren JF, Tschirhart J, Anderson T, Ando AW, Beissinger SR, et al. 1999. Why economics matters for endangered species protection. *Conserv. Biol.* 13(6):1257–61
Smith JA, Tommasi D, Sweeney J, Brodie S, Welch H, et al. 2020. Lost opportunity: quantifying the dynamic economic impact of time-area fishery closures. *J. Appl. Ecol.* 57(3):502–13
Smith MD, Lynham J, Sanchirico JN, Wilson JA. 2010. Political economy of marine reserves: understanding the role of opportunity costs. *PNAS* 107(43):18300–5
Smith MD, Wilen JE. 2003. Economic impacts of marine reserves: the importance of spatial behavior. *J. Environ. Econ. Manag.* 46(2):183–206
Smith MD, Wilen JE. 2004. Marine reserves with endogenous ports: empirical bioeconomics of the California sea urchin fishery. *Mar. Resour. Econ.* 19(1):85–112
Smith MD, Zhang J, Coleman FC. 2006. Effectiveness of marine reserves for large-scale fisheries management. *Can. J. Fish. Aquat. Sci.* 63(1):153–64
Spalding M, Burke L, Wood SA, Ashpole J, Hutchison J, zu Ermgassen P. 2017. Mapping the global value and distribution of coral reef tourism. *Mar. Policy* 82:104–13
Stefanski SF, Shimshack JP. 2016. Valuing marine biodiversity in the Gulf of Mexico: evidence from the proposed boundary expansion of the Flower Garden Banks National Marine Sanctuary. *Mar. Resour. Econ.* 31(2):211–32
Sultan R. 2021. Fisheries benefits of a marine protected area with endogenous fishing efforts—a bioeconomic analysis. *Ocean Coast. Manag.* 207:105594
Sultan RM. 2020. Modelling fishing location choice and spatial behaviour of fishers near a marine protected area. *Mar. Econ. Manag.* 4(1):1–22
Sutcliffe PR, Klein CJ, Pitcher CR, Possingham HP. 2015. The effectiveness of marine reserve systems constructed using different surrogates of biodiversity. *Conserv. Biol.* 29(3):657–67
Swallow SK, Wear DN. 1993. Spatial interactions in multiple-use forestry and substitution and wealth effects for the single stand. *J. Environ. Econ. Manag.* 25:103–20

Teixeira JB, Moura RL, Mills M, Klein C, Brown CJ, et al. 2018. A habitat-based approach to predict impacts of marine protected areas on fishers. *Conserv. Biol.* 32(5):1096–1106

Thiault L, Kernaléguen L, Osenberg CW, Lison de Loma T, Chancerelle Y, et al. 2019. Ecological evaluation of a marine protected area network: a progressive-change BACIPS approach. *Ecosphere* 10(2):e02576

Vandeperre F, Higgins RM, Sánchez-Meca J, Maynou F, Goñi R, et al. 2011. Effects of no-take area size and age of marine protected areas on fisheries yields: a meta-analytical approach. *Fish Fish*. 12(4):412–26

Velando A, Munilla I. 2011. Disturbance to a foraging seabird by sea-based tourism: implications for reserve management in marine protected areas. *Biol. Conserv.* 144(3):1167–74

Vilas D, Coll M, Corrales X, Steenbeek J, Piroddi C, et al. 2020. The effects of marine protected areas on ecosystem recovery and fisheries using a comparative modelling approach. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 30(10):1885–1901

Voorberg W, Van der Veer R. 2020. Co-management as a successful strategy for marine conservation. *J. Mar. Sci. Eng.* 8(7):491

Waldron A, Adams V, Allan J, Arnell A, Asner G, et al. 2020. *Protecting 30% of the planet for nature: costs, benefits and economic implications*. Work. Pap., Campaign Nat., Washington, DC. **http://hdl.handle.net/10138/326470**

Waldron A, Miller DC, Redding D, Mooers A, Kuhn TS, et al. 2017. Reductions in global biodiversity loss predicted from conservation spending. *Nature* 551(7680):364–67

Walters C, Hilborn R, Costello C. 2008. *Comparison of marine protected area policies using a multispecies, multigear equilibrium optimization model (EDOM)*. Work. Pap., Fish. Cent., Vancouver, BC. **http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.571.7431&rep=rep1&type=pdf**

Welch H, Pressey RL, Reside AE. 2018. Using temporally explicit habitat suitability models to assess threats to mobile species and evaluate the effectiveness of marine protected areas. *J. Nat. Conserv.* 41:106–15

Wilhelm TA, Sheppard CRC, Sheppard ALS, Gaymer CF, Parks J, et al. 2014. Large marine protected areas—advantages and challenges of going big. *Aquat. Conserv. Mar. Freshw. Ecosyst.* 24(S2):24–30

Wilson KL, Tittensor DP, Worm B, Lotze HK. 2020. Incorporating climate change adaptation into marine protected area planning. *Glob. Change Biol.* 26(6):3251–67

Yamazaki S, Hoshino E, Resosudarmo B. 2015. No-take marine reserves and illegal fishing under imperfect enforcement. *Aust. J. Agric. Resour. Econ.* 59(3):334–54

Zhang J, Smith MD. 2011. Heterogeneous response to marine reserve formation: a sorting model approach. *Environ. Resour. Econ.* 49(3):311–25