Ann. N.Y. Acad. Sci. ISSN 0077-8923

ANNALS OF THE NEW YORK ACADEMY OF SCIENCES
Issue: *The Year in Ecology and Conservation Biology*
REVIEW

# A novel framework for analyzing conservation impacts: evaluation, theory, and marine protected areas

Michael B. Mascia,[1] Helen E. Fox,[2] Louise Glew,[3] Gabby N. Ahmadia,[3] Arun Agrawal,[4] Megan Barnes,[5,6] Xavier Basurto,[7] Ian Craigie,[8] Emily Darling,[9,10] Jonas Geldmann,[11] David Gill,[12,13,a] Susie Holst Rice,[14] Olaf P. Jensen,[15] Sarah E. Lester,[16] Patrick McConney,[17] Peter J. Mumby,[18] Mateja Nenadovic,[7] John E. Parks,[19] Robert S. Pomeroy,[20] and Alan T. White[21,b]

[1]Betty and Gordon Moore Center for Science, Conservation International, Arlington, Virginia. [2]National Geographic Society, Washington, DC. [3]World Wildlife Fund–US, Washington, DC. [4]School of Natural Resources & Environment, University of Michigan, Ann Arbor, Michigan. [5]Natural Resources and Environmental Management, University of Hawaii at Manoa, Honolulu, Hawaii. [6]Centre for Excellence in Environmental Decisions, University of Queensland, Brisbane, Queensland, Australia. [7]Nicholas School of the Environment, Duke University, Beaufort, North Carolina. [8]ARC Center of Excellence for Coral Reef Studies, James Cook University, Townsville, Queensland, Australia. [9]Department of Ecology and Evolutionary Biology, University of Toronto, Toronto, Ontario, Canada. [10]Marine Program, Wildlife Conservation Society, Bronx, New York. [11]Conservation Science Group, Department of Zoology, University of Cambridge, Cambridge, UK; and Center for Macroecology, Evolution, and Climate, Natural History Museum of Denmark, University of Copenhagen, Copenhagen, Denmark. [12]National Socio-Environmental Synthesis Center (SESYNC), Annapolis, Maryland. [13]Luc Hoffmann Institute, WWF International, Gland, Switzerland. [14]Coral Reef Conservation Program, National Oceanic and Atmospheric Administration, Silver Spring, Maryland. [15]Institute of Marine & Coastal Sciences, Rutgers University, New Brunswick, New Jersey. [16]Department of Geography, Florida State University, Tallahassee, Florida. [17]Centre for Resource Management and Environmental Studies (CERMES), University of the West Indies, Cave Hill, Barbados. [18]Marine Spatial Ecology Lab, School of Biological Sciences, The University of Queensland, Brisbane, Queensland, Australia. [19]Marine Management Solutions, Honolulu, Hawaii. [20]Department of Agricultural and Resource Economics, University of Connecticut-Avery Point, Groton, Connecticut. [21]Asia-Pacific Program, The Nature Conservancy, Honolulu, Hawaii

Address for correspondence: Michael B. Mascia, Betty and Gordon Moore Center for Science, Conservation International, 2011 Crystal Drive, Suite 500, Arlington, VA 22202. mmascia@conservation.org

**Environmental conservation initiatives, including marine protected areas (MPAs), have proliferated in recent decades. Designed to conserve marine biodiversity, many MPAs also seek to foster sustainable development. As is the case for many other environmental policies and programs, the impacts of MPAs are poorly understood. Social–ecological systems, impact evaluation, and common-pool resource governance are three complementary scientific frameworks for documenting and explaining the ecological and social impacts of conservation interventions. We review key components of these three frameworks and their implications for the study of conservation policy, program, and project outcomes. Using MPAs as an illustrative example, we then draw upon these three frameworks to describe an integrated approach for rigorous empirical documentation and causal explanation of conservation impacts. This integrated *three-framework* approach for *impact evaluation* of *governance* in *social–ecological systems* (3FIGS*)* accounts for alternative explanations, builds upon and advances social theory, and provides novel policy insights in ways that no single approach affords. Despite the inherent complexity of social–ecological systems and the difficulty of causal inference, the 3FIGS approach can dramatically advance our understanding of, and the evidentiary basis for, effective MPAs and other conservation initiatives.**

**Keywords:** protected areas; impact evaluation; social–ecological systems; common-pool resources; governance; biodiversity conservation; ecological integrity; human well-being

[a]Current affiliation: Betty and Gordon Moore Center for Science, Conservation International, Arlington, VA

[b]Current affiliation: USAID Sustainable Ecosystems Advanced (SEA) Project, Tetratech, Jakarta, Indonesia

doi: 10.1111/nyas.13428

## Introduction

Recent decades have witnessed rapid growth in the diversity, abundance, and spatial extent of environmental conservation initiatives designed to conserve biodiversity and foster sustainable development.[1–3] National parks, hunting regulations, endangered species legislation, pollution control laws, and other traditional state-centered regulatory policies have been joined by formal systems of payments for ecosystem services,[4] community-based management,[5,6] environmental certification,[7] and natural resource privatization (e.g., privately protected areas, conservation easements, and catch shares).[8–10] Despite the growing number and diversity of conservation initiatives, the impacts of these interventions are often only poorly understood.[11,12]

Marine protected areas (MPAs), a cornerstone of global efforts to conserve marine biodiversity and alleviate coastal poverty, typify the proliferation of conservation initiatives. Also known as marine reserves, sanctuaries, managed areas, and parks, an MPA is "any area of the intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment."[13] The number and spatial extent of MPAs have increased severalfold since 1990, accelerating in recent years, such that more than 12,000 MPAs now cover 8.4% of coastal waters and 3.4% of the global ocean.[14,15] MPAs vary dramatically in size, regulatory restrictions, governance, and social and ecological contexts.[16–18] MPAs are established for diverse reasons,[19,20] often with multiple and sometimes conflicting goals: small, locally managed MPAs are often designed to enhance food security and manage data-poor fisheries,[18] while large "Big Ocean" MPAs contribute to full representation of marine habitats and progress toward Convention on Biological Diversity targets for MPAs to cover 10% of the global ocean by 2020.[14,15,21]

The ecological impacts of MPAs are highly variable, ranging from positive to negative.[22–24] MPA establishment often increases the density, biomass, size, and diversity of otherwise-exploited species within MPA boundaries,[23–25] with subsequent indirect impacts on other species through cascading ecological interactions (e.g., parrotfish increases, macroalgae declines, and subsequent coral recovery).[26–28] Available evidence, although limited, suggests that MPAs that prohibit fishing can lead to enhanced ecological resilience, including resilience to climate fluctuations,[29,30] storm disturbance,[31] and invasive species.[32] Corresponding impacts in areas adjacent to MPAs include increased larval recruitment,[33] spillover of adult fish,[34,35] and reduced fear behaviors among fish.[36] The wide variation in direction and magnitude of MPA ecological impacts has been attributed to MPA governance (e.g., MPA age, size, resource-use rules, and compliance), management capacity (e.g., staff and budget),[24] biophysical factors (e.g., isolation by deep water), and species traits (e.g., range or life history characteristics).[37–40] A significant portion of the variability in ecological outcomes remains unexplained.[23,24,41]

The social impacts of MPAs are widely debated and less well understood than their ecological analogues.[22,42] The intended and unintended social impacts of MPAs vary across social domains,[42,43] spatial and temporal scales,[44] and within and among social groups.[42,45] MPA establishment reallocates property rights[46] over marine resources, thus restructuring relationships among stakeholders and transforming spatial and temporal patterns of fishing and other activities.[47,48] In many cases, reallocation of MPA resource rights affects multiple dimensions of human well-being[42,49] through processes mediated by the characteristics of individual resource users (e.g., gear type)[42,44] and contextual characteristics (e.g., human population size).[50] The resulting MPA impacts on economic well-being, cultural identity, and social conflict can be positive, neutral, or negative (e.g., Refs. 43 and 51–53). Evidence suggests that MPAs significantly increase aspects of food security for most fisher subgroups,[42] but the long-term implications of MPAs for other health outcomes remain largely unexplored.[54,55] The magnitude and direction of MPA social impacts are frequently dependent on scale, varying at different levels of social organization.[44] Social impacts also vary in their degree of permanence,[43] with some more likely to manifest almost immediately (e.g., resource rights)[47] and others emerging over longer timescales (e.g., child nutrition).[54] MPAs

The copyright line for this article was changed on July 29, 2017 after original online publication.

frequently confer economic benefits and costs to those who use MPA resources; these benefits and costs are often unequally distributed among individuals, among social groups, and across space and time.[47,56–58] MPA economic benefits may include increased revenue from nonextractive uses and the capture of nonuse value (e.g., donations and user fees).[59–62] Conversely, extractive users may face significant opportunity and financial costs from MPA establishment.[58,63]

Scholars and practitioners have proposed diverse analytic frameworks for monitoring and evaluating MPAs (e.g., Refs. 64–66) and analogous conservation initiatives (e.g., Refs. 67 and 68), but three distinct approaches have gained currency: social–ecological systems (SESs), impact evaluation, and common-pool resource governance.[69–71] An SES is an integrated system characterized by complex, interdependent, and reciprocal relationships between human society and ecosystems.[72] Impact evaluation, by contrast, is the systematic process of assessing the causal effects of a project, program, or policy.[73] Common-pool resource governance refers to the system of decision-making arrangements, resource-use rights, monitoring and enforcement systems, and conflict-resolution mechanisms that shape human interactions with common-pool natural resources. To date, the relationships among these three approaches remain relatively unexplored.

To further scientific efforts to document and explain variation in the impacts of conservation initiatives, we review the relationships among key components of SESs, impact evaluation, and common-pool resource governance. In particular, we examine the SES framework for classifying social–ecological variables, the research design tenets of impact evaluation, and hypotheses derived from theories of common-pool resource governance. Using MPAs as an illustrative example, we then draw upon these three approaches to describe an integrated approach for rigorous empirical documentation and causal explanation of conservation impacts. This integrated *three-framework* approach to *impact evaluation* of *governance* in *social–ecological systems* (3FIGS) controls for alternative causal explanations, builds upon and advances social theory, and provides novel policy insights in ways that no single approach affords. We conclude with a discussion of the merits and limits of 3FIGS for evaluation of MPAs and other conservation interventions.

## 3FIGS conceptual foundations

### *Social–ecological systems framework*

The SES framework is a hierarchical classification system that helps to "identify the basic working parts and critical relationships" that characterize coupled social and ecological systems.[74] As part of the broader literature on SESs,[75–77] the SES framework "facilitate[s] multidisciplinary efforts toward a better understanding of complex SESs"[78]—particularly those involving common-pool natural resources.[74] At its most basic, the SES framework categorizes the relationships and interactions (Fig. 1; I) among specific actors (A), governance systems (GS), natural resource systems (RS), and resource units (RU)—and the social and ecological outcomes (O) that flow from these interactions. The interactions among these four subsystems (A, GS, RS, and RU) and the outcomes (O) that result from them are the products of a *focal action situation* (a situation in which inputs are transformed by multiple actors into outcomes);[74] these subsystems, interactions, and outcomes are embedded within a broader social (S) and ecological context (ECO) that may shape—and be shaped by—the SESs.[78] Each subsystem comprises a specific set of attributes (Table 1), each of which can be further disaggregated.[74,78,79] While higher level attributes are universal to any SES, lower level attributes are specific to particular systems.[79,80]

The SES framework provides a heuristic tool for characterizing conservation interventions and their impacts.[77] Conservation interventions are policies, programs, and projects—often novel governance systems—designed to shape the behavior of specific actors and thus conserve natural resources.[81] These interactions are embedded within a broader social and ecological context that shapes—and is shaped by—the performance of the conservation intervention.[16] Thus, rearranging the SES subsystems can highlight the relationships between a governance-focused conservation intervention and its impacts (Fig. 2). As a classificatory system, however, the SES framework does not provide guidance regarding the hypothesized causal relationships among variables or the research designs that one might employ to examine these relationships.[74]

### *Impact-evaluation research design*

Impact evaluation places particular emphasis on research designs that permit causal inference.[11,73] Causal inference rests on the comparison of

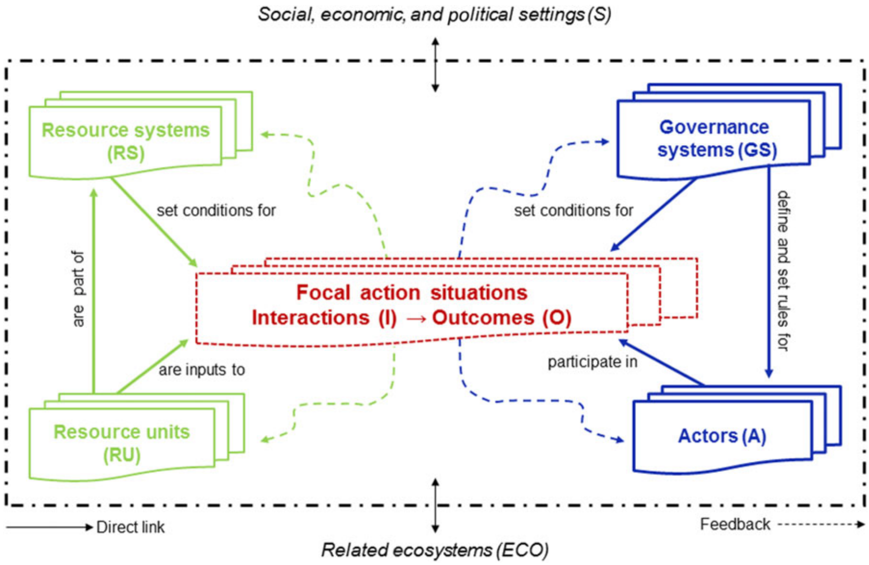


**Figure 1.** The first-tier variables within a social–ecological system (SES). "The subsystems are (i) resource systems (e.g., a designated protected park encompassing a specified territory containing forested areas, wildlife, and water systems), (ii) resource units (e.g., trees, shrubs, and plants contained in the park, types of wildlife, and amount and flow of water), (iii) governance systems (e.g., the government and other organizations that manage the park, the specific rules related to the use of the park, and how these rules are made), and (iv) users (e.g., individuals who use the park in diverse ways for sustenance, recreation, or commercial purposes). Each core subsystem is made up of multiple second-level variables (e.g., size of a resource system, mobility of a resource unit, level of governance, users' knowledge of the resource system) . . . which are further composed of deeper level variables."[68] Used with permission from McGinnis and Ostrom;[74] adapted from Ref. 68.

outcomes observed under an intervention with an estimate of what would have happened in the absence of that intervention (i.e., the counterfactual);[82] this comparison provides an estimate of the intended and unintended impacts attributable to an intervention.[73,82] Causal relationships between an intervention and its impacts can be explored with experiments (where random assignment to treatment and nontreatment groups is feasible)[82] or, more commonly in SESs, quasiexperiments that apply statistical techniques (e.g., regression discontinuity, matched comparisons, and synthetic controls)[73,82] to control for systematic differences between treated (e.g., MPA) and untreated (e.g., non-MPA) groups. Other research designs are also employed in impact evaluation (e.g., in-depth case studies), "though the credibility of their estimates of program effects relies on how well the studies' designs rule out competing causal explanations."[83]

Central to impact evaluation is the exclusion of alternative explanations for the outcomes observed in intervention sites.[84] Impact evaluations control for factors that may bias the magnitude or direction of impact estimates (e.g., attributes of the actors (A), resource systems (RS), resource units (RU), and social and ecological context (S; ECO)),[85] thus creating analyses of paired SESs that vary only in their governance and, perhaps, in their outcomes (Fig. 3). All other subsystems (and elements) of the SES framework (e.g., GS) represent variables that researchers must control for (e.g., through randomization or statistical matching) or explicitly incorporate into analyses as explanatory variables. Where SESs processes in an intervention influence the status of outcome variables (e.g., biomass and economic well-being) in nonintervention sites, appropriate impact evaluation design must also employ methods to eliminate bias arising from either leakage (e.g., the displacement of harvest effort from intervention to nonintervention sites)[86] or spill-over effects (e.g., increased harvest in nonintervention sites due to biomass increases generated by intervention).[51] At their simplest, comparative analyses of these paired SESs allow researchers to document the impacts of an

**Table 1. The second-tier variables of a social–ecological system (SES)**[74]

| First-tier variable | Second-tier variable |
|---|---|
| Social, economic, and political settings (S) | S1—Economic development |
| | S2—Demographic trends |
| | S3—Political stability |
| | S4—Other governance trends |
| | S5—Market stability |
| | S6—Media organizations |
| | S7—Technology |
| Resource systems (RS) | RS1—Sector (e.g., water, pasture, forests, and fish) |
| | RS2—Clarity of system boundaries |
| | RS3—Size of resource systems |
| | RS4—Human-constructed facilities |
| | RS5—Productivity of system |
| | RS6—Equilibrium properties |
| | RS7—Predictability of system dynamics |
| | RS8—Storage characteristics |
| | RS9—Location |
| Governance systems (GS) | GS1—Government organizations |
| | GS2—Nongovernmental organizations |
| | GS3—Network structure |
| | GS4—Property rights systems |
| | GS5—Operational-choice rules |
| | GS6—Collective-choice rules |
| | GS7—Constitutional-choice rules |
| | GS8—Monitoring and sanctioning rules |
| Resource units (RU) | RU1—Resource unit mobility |
| | RU2—Growth or replacement rate |
| | RU3—Interaction among resource units |
| | RU4—Economic value |
| | RU5—Number of resource units |
| | RU6—Distinctive characteristics |
| | RU7—Spatial and temporal characteristics |
| Actors (A) | A1—Number of relevant actors |
| | A2—Socioeconomic attributes |
| | A3—History or past experiences |
| | A4—Location |
| | A5—Leadership/entrepreneurship |
| | A6—Norms (trust-reciprocity)/social capital |
| | A7—Knowledge of SES/mental models |
| | A8—Importance of resource (dependence) |
| | A9—Technologies available |
| Action situations: interactions (I)–outcomes (O) | I1—Harvesting |
| | I2—Information sharing |
| | I3—Deliberation processes |
| | I4—Conflicts |
| | I5—Investment activities |
| | I6—Lobbying activities |
| | I7—Self-organizing activities |
| | I8—Networking activities |
| | I9—Monitoring activities |
| | I10—Evaluative activities |
| | O1—Social performance measures (e.g., efficiency, equity, sustainability, and accountability) |
| | O2—Ecological performance measures (e.g., overharvested, resilience, biodiversity, and sustainability) |
| | O3—Externalities to other SESs |
| Related ecosystems (ECO) | ECO1—Climate patterns |
| | ECO2—Pollution patterns |
| | ECO3—Flows into and out of focal SESs |

*Continued*

**Table 1.** *Continued*

Note: Adapted from Ref. 78.

intervention (e.g., Refs. 87 and 88), while emerging approaches highlight the potential to explore the mechanisms and pathways by which impacts occur (e.g., Ref. 89), variation in impacts within and among groups, the attributes of an intervention that foster positive (or negative) impacts, and the contexts in which an intervention is most likely to succeed or fail.[12,90] The SES framework provides a valuable heuristic for researchers attempting to identify and control for systematic biases that might confound the relationship between interventions and outcomes, precluding causal inference. Substantial barriers to understanding the impact of interventions on SESs remain, however, given that impact evaluation provides no guidance regarding the specific variables to measure in a given SES or the hypothesized relationships among them.[91]

### *Common-pool resource governance theory*

Theories of common-pool resource governance[92–94] seek to explain the emergence, evolution, and performance of regimes governing common-pool natural resources and analogous systems. In common pool resources, use is *rival* (i.e., use by one individual diminishes the resources available to another) and *exclusion* of potential resource users is difficult or costly, creating the potential for overexploitation of scarce resources.[95,96] Theories of common-pool resource governance have highlighted four

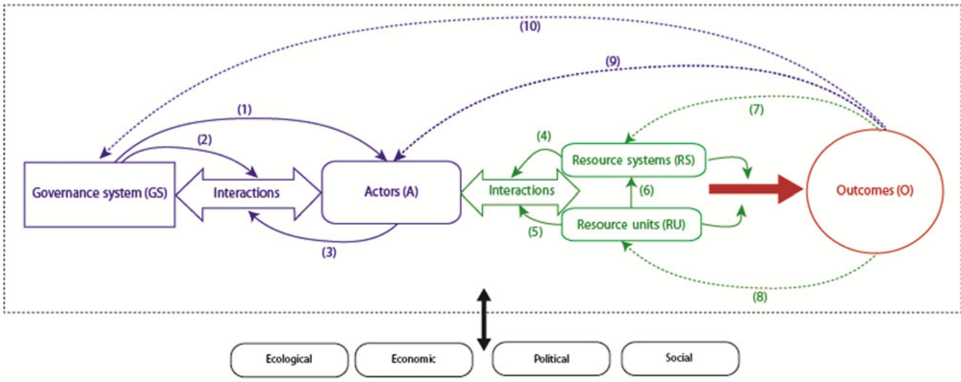


**Figure 2.** The first-tier variables within a social–ecological system (SES), reorganized to emphasize the causal relationships between governance systems and outcomes. Conservation interventions often represent novel governance systems that induce a causal chain of events with intended and unintended consequences, feedback loops, synergies, and trade-offs. Related ecosystems (ECO) and social, economic, and political settings (S) in Figure 1 are collectively represented here as contextual variables. Solid lines represent direct links; dashed lines represent feedback mechanisms. Numbered arrows represent links labeled in Figure 1 as (1) "set rules for," (2) "set conditions for," (3) "participate in," (4) "set conditions for," (5) "are inputs to," and (6) "are part of." Numbered arrows (7–10) represent feedback mechanisms.

dimensions of governance that profoundly affect the performance of regimes governing fisheries, forests, grasslands, and other common-pool resources: decision-making arrangements (Table 1; GS6, GS7), resource-use rights (GS4, GS5), monitoring and enforcement systems (GS8), and conflict-resolution mechanisms.[78,92] In each of these four dimensions, research has identified variables hypothesized to shape outcomes in specific ways (Table 2). Theories of common-pool resource governance also provide insights into the diverse factors that foster the emergence and evolution of governance regimes through collective action,[92] highlighting the complex relationships among different components of the SES framework.[78] Indeed, many elements of the SES framework have their origins in the early analytic frameworks of the literature on common-pool resource governance.[77,97]

Originally developed in the context of small-scale common-pool resource systems,[92,98,99] theories of common-pool resource governance are valid starting hypotheses for predicting impacts of conservation interventions in larger and more complex SESs.[77,78] Ostrom's design principles for sustainable regimes,[92,93] for example, are hypotheses to operationalize and test within an impact-evaluation research design (Fig.4). Though the large number of potential explanatory variables complicates hypothesis testing,[94] and reductionist methods may obscure emergent properties,[74,79] careful attention to research design can help to address these challenges.[77,97] Thus, theories of common-pool resource governance have the potential to catalyze a substantive advance in the rigor and reach of impact evaluations.[91] Likewise, rigorous research designs informed by impact evaluation may foster important insights into common-pool resource governance theory.

## Operationalizing 3FIGS: MPAs as an illustrative example

MPAs illustrate the potential for an integrated three-framework approach to influence evaluation of common-pool resource governance in SESs. MPAs are an exemplar of an SES[69,78] increasingly examined through the lens of impact evaluation[22,43,100] and the theories of common-pool resource governance.[16,101] However, like studies of other conservation interventions, MPA research has yet to fully integrate the research questions, methods, and study designs of impact evaluation; the principles and hypotheses of common-pool resource governance theory; and the conceptual framework of SESs. Here, we outline the potential for 3FIGS to provide an integrated approach to examining the impacts of conservation interventions, drawing upon the SES framework (Fig. 1) to identify (1) the potential ecological and social outcomes of an intervention and (2) the confounding ecological and social processes to be controlled to obtain

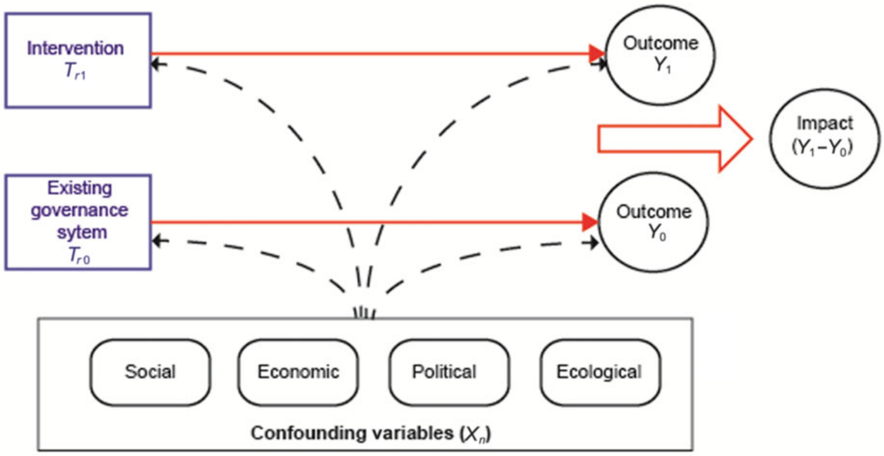


**Figure 3.** Directed acyclic graph (DAG) illustrating how the essential social–ecological relationships may be examined through the lens of impact evaluation. Contextual variables in Figure 2 are illustrated here as confounding variables that are explicitly controlled through research design. We use the term *impact* to denote a generic treatment effect (i.e., the causal impact of an intervention on a variable of interest).

an unbiased estimate of MPA impacts. Having identified potential outcome variables, impact-evaluation research designs provide a methodology for documenting MPA impacts while controlling for confounding SES interactions and processes (Fig. 3). Finally, elements of common-pool resource governance, nested as differential treatments within an impact-evaluation research design, allow an examination of the mechanisms by which MPAs have ecological and social impacts (Fig. 4).

### *MPA governance systems*[c]

Governance constitutes the processes by which authority is conferred, by which authoritative decisions are made, and by which these decisions are enforced and modified.[102–104] Governance is a principal component of SESs,[78] structuring human interactions across all levels of social organization (Fig. 2).[104,105] Like many conservation interventions, MPAs are, essentially, novel governance systems that explicitly or implicitly define *who* may do *what*—and *where*, *when*, and *how* they may do it—with respect to a specific, spatially bounded portion of the environment (denoted by blue elements in Fig. 2).[16] In many cases, MPAs are layered upon existing marine governance systems, modifying or replacing the prior governance regime.

The four principal components of MPA governance (decision-making arrangements, resource-use rules, monitoring and enforcement systems, and conflict-resolution mechanisms) directly and indirectly shape human resource-use patterns and, ultimately, the ecological and social impacts of MPAs.[16] Each of these four elements may have both formal and informal components derived from diverse sources, including legal statutes, policy statements, judicial decisions, organizational practices, social norms, and cultural traditions. As a result, the *de facto* system of governance that actually governs an MPA often differs sharply from the *de jure* regime established through formal legal structures and policy processes.[16] The *de facto* system of rules and the processes through which these rules are developed, implemented, and adapted over time significantly influence MPA impacts. Specific governance characteristics—known as *design principles*—are associated with long-term sustainability in institutions governing common-pool resources,[92] such as conservation interventions.[22,106]

MPA decision-making arrangements specify the rights of individuals or groups to make choices regarding other aspects of MPA design and management. Decision-making arrangements shape rules governing resource use by defining the interests represented and manifested in policymaking processes. Collective-choice rules (Table 1; GS6) determine, for example, who may participate in making decisions

[c]This section is adapted from and builds upon Ref. 16.

**Table 2. Domains of marine protected area (MPA) governance, hypothesized attributes of successful MPA governance, and illustrative indicators to monitor**[16,22,130]

| Domain | Hypothesized attribute | Illustrative indicator |
|---|---|---|
| Decision-making arrangements | Resource users participate in decision making | Proportion of user groups participating in decision to establish MPA |
| | Resource user may self-govern resource use | Proportion of users whose right to self-organize is minimally recognized by government |
| Resource-use rights | Resource users defined clearly | Proportion of important users subject to specific appropriation rule |
| | Resources defined clearly | Proportion of MPA boundary clearly demarcated to users |
| | Resource user costs and benefits roughly proportional | MPA cost–benefit ratio to resource users |
| | Resource user rights linked to local conditions | Proportion of MPA rules context dependent |
| Monitoring and enforcement systems | Monitors assess resource conditions | Ecological monitoring patrol frequency |
| | Monitors assess user behavior | Compliance monitoring patrol frequency |
| | Monitors are resource users or are accountable to users | Sanctions for failing to monitor resource |
| | Sanctions for noncompliance likely | Frequency of sanctions for noncompliance |
| | Sanctions for noncompliance graduated | Mean types of sanctions levied for noncompliance |
| | Sanctions for noncompliance context dependent | Mean number of contextual factors considered when imposing sanction for noncompliance |
| Conflict-resolution mechanisms | Conflict-resolution mechanisms accessible to users | Mean time/cost to resolve conflict |

Note: Domains, hypothesized attributes, and illustrative indicators derived from theories of common-pool resource governance.[92,93]

and who may not (e.g., government officials and resource users), how decision makers are selected for their positions (e.g., elected or appointed), and how decisions are made (e.g., consensus or majority vote).[92] Constitutional-choice arrangements (GS7) define how decisions are made regarding the configuration of collective-choice arrangements;[92] these higher order social institutions shape both MPA governance and impacts by defining how site-level rules are made and who makes them.[16] Subtle differences in the rules that govern MPA decision making may have significant impacts on MPA design, implementation, and evaluation (e.g., U.S. National Marine Sanctuaries).[107] Governance regimes that recognize the rights of resource users to participate in decision making and to self-govern tend to be more effective than those that centralize authority in a few individuals or actors (Table 2).[92]

Rules governing resource use (Table 1; GS4, GS5) are the second principal element of MPA governance. Resource-use rules (denoted by arrow 1 in Fig. 2)—including laws, regulations, formal and informal policies, codes of conduct, and social norms—specify the rights of individuals or groups to access and appropriate resources. Held by individuals, groups, organizations, or the state, these rights establish standards for interactions among individuals and between individuals and the marine environment ("actors" and "interactions" in Fig. 2). Infinite possible configurations of resource-use rules exist, ranging along a continuum from "open access" (i.e., no rules/any and all uses) to a complete prohibition on human activities. Rules may apply to specific locations, times, resources, and actors. Individuals or groups may be defined by actors' identities, geographic proximity, tenure, activity type (e.g., consumptive versus nonconsumptive), intent (e.g., targeted species, recreational versus commercial activity), or practices (e.g., fishing gear). Consequently, the rules governing marine resources may shape the direction, magnitude, and distribution of MPA impacts. Four attributes

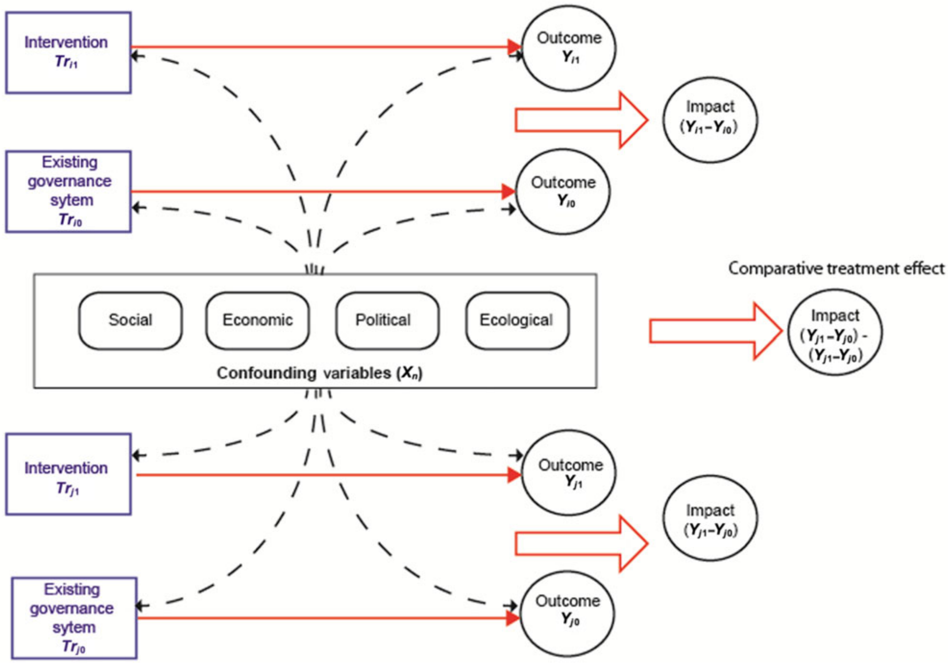


**Figure 4.** Alternative hypotheses from theories of common-pool resource governance examined through the lens of impact evaluations. The relative treatment effect of paired conservation interventions, representing the single null ($Tr_i$) or alternative common-pool resource hypotheses ($Tr_j$) (e.g., protected area without active and accountable monitoring versus protected area with active and accountable monitoring), can be examined through impact evaluation. Contextual variables in Figure 2 are illustrated here as confounding variables that are explicitly controlled through research design. We use the term *impact* to denote a generic treatment effect (i.e., the causal impact of an intervention on a variable of interest). "Comparative impact" refers to the net difference in treatment effects between conservation interventions *i* and *j*.

of resource-use rights are associated with sustainable governance systems: (1) clearly defined resource users, (2) clearly defined resources, (3) contextually appropriate resource-use rights, and (4) roughly proportional impacts on resource users.[92,93] Unclear boundaries may limit the ability of resource users to develop appropriate rules and monitor compliance. Incongruent rules or those with disproportionate effects on particular social or economic groups may undermine compliance.[92] Ecologists also note that MPA age, size, location, spatial configuration, and fishing prohibitions—all manifestations of resource-use rights—often shape MPA impacts (Table 2) (e.g., Refs. 38, 39, and 41).

MPA monitoring systems (Table 1; I9, GS8) seek to track changes in the state of MPA-associated social and environmental systems. MPA monitoring systems vary in what they measure and who does the measuring, as well as where, when, and how measurements are made. Monitoring can provide decision makers with insights on social and ecological conditions (ambient monitoring), track the progress of management actions toward a specified goal (performance measurement), or document the intended or unintended impacts of MPA establishment (impact evaluation).[90] In addition, the involvement of resource users and other nonscientists in formal data collection and analysis (i.e., participatory monitoring) may also provide a mechanism for increasing awareness, improving resource management, and empowering communities.[108] In practice, relatively few MPAs have comprehensive monitoring systems; resource users, managers, and other stakeholders often informally monitor environmental and social indicators to assess MPA performance (Table 2).[16]

Sanctioning systems (Table 1; GS8) attempt to increase compliance with rules governing resource use by monitoring user behavior (I9) and punishing those engaged in prohibited activities. By increasing the severity and likelihood of sanctions and thus raising the opportunity cost of noncompliance, enforcement systems act directly on resource users to foster adherence to established rules. Research highlights six attributes of monitoring and sanctioning systems that are hypothesized to enhance performance of MPAs and other natural resource governance regimes: monitoring systems that assess (1) resource conditions and (2) user behavior; and (3) ensure accountability to users; and enforcement systems with (4) meaningful but (5) graduated and (6) context-dependent sanctions, which ensure that the punishment fits the crime (Table 2).[92]

Lastly, conflict-resolution mechanisms are formal and informal processes for resolving disputes. Conflict-resolution mechanisms permit information exchange, clarification of resource-use rights, and adjudication of disputes related to decision making, resource use, monitoring, and enforcement. Though not highlighted in the SES framework, readily accessible and low-cost conflict-resolution mechanisms enhance regime performance by mitigating social conflict and thereby minimizing resource overexploitation and dissipation of benefits from common-pool resources.[92] Conflict-resolution mechanisms also enhance MPA performance by giving voice to aggrieved parties and acknowledging their concerns, which increases the legitimacy of MPA rules and regulations. Evidence suggests that (1) rapid, (2) low-cost, and (3) easily accessible conflict-resolution mechanisms are more likely to facilitate negotiation between actors than more complex, expensive, or remote systems (Table 2).[92]

### *Resource-use patterns: actors and their interactions*

MPA governance regimes (Fig. 1; GS) directly and indirectly structure interactions (I) between humans (A) and marine species (RU) and ecosystems (RS) by defining the rules governing resource use and by shaping rates of compliance with these rules. These patterns of consumptive and nonconsumptive resource access, use, and appropriation (shown in blue in Fig. 2; I1) may be characterized by five sets of variables: the resources used; demographic attributes of the users; location of use; timing of use; and mode of use.[109] (In other words, *what* is being used *by whom, where, when,* and *how*?) The demographics of resource use—*who* uses resources—can be characterized by both the number of users (A1) and their social attributes (A2; e.g., gender, age, class, place of residence, education, culture, ethnicity, and religion), though not all of these variables will be salient in all cases. Actors' past experiences (A3), mental models (A7), and degree of resource dependence (A8) also shape resource-use patterns. Location (A4) and timing of use also define resource-use patterns in simple but important ways. Lastly, the mode (A9) and intensity of resource use, including both consumptive (e.g., fishing) and nonconsumptive practices (e.g., dive tourism), are key characteristics of actors' resource-use patterns.[110] Resource-use patterns may vary in any one or all of these dimensions; these differences may lead to significant differences in the biological and social impacts of MPAs.[37,109]

### *Resource systems, units, and interactions*

Resource-use patterns are structured by characteristics of resource systems and resource units (as illustrated by green elements in Fig. 2). Most wild marine plants and animals (Table 1; RS1) are common-pool resources characterized by a high degree of rivalry and by difficulty of excluding others,[92] though some marine species and ecosystems have distinctive characteristics (RU6) and clear boundaries (RS2) that facilitate exclusion. The abundance (RU5), economic value (RU4), and spatial and temporal distribution (RS3, RS9, and RU7) of marine resources all influence the likelihood of (un)sustainable use. Similarly, resource productivity (RS5) and robustness (depletability; RU2)—the ability of a resource to sustain itself in the face of human use (e.g., based on population growth rates)—differentiate marine species and ecosystems from each other.[111] Resource mobility (i.e., spatial movement of the resource, independent of user actions; RU1) and the capacity for storage (i.e., to capture and retain the resource for later use; RS8) are also key attributes that shape how humans govern and use marine resources.[112] Additional biological attributes of living marine resources (e.g., species, size, sex, age, and behavior) may shape how people interact with the marine environment and the impacts of these interactions.

### *MPA outcomes*

Both theory and empirical evidence suggest that resource attributes, human resource-use patterns, and local context (discussed below) influence the ecological conditions (Table 1; O2) within and adjacent to an MPA. Common population-level

**Table 3. Ecological domains, attributes, and illustrative indicators of the ecological impacts of marine protected areas (MPAs)**[113–115]

| Domain | Attribute | Illustrative indicator |
|---|---|---|
| Population | Numeric density | Count of individuals per unit area |
| | Biomass | Abundance multiplied by mean weight |
| | Age structure | Frequency distributions of age classes |
| | Recruitment | Count of larvae per unit area |
| Community | Species richness | Number of species per survey |
| | Species diversity | Simpson diversity index |
| | Trophic structure | Distribution of trophic levels in species assemblage |
| | Functional redundancy | Number of species within functional groups |
| Ecosystem | Productivity | Fishery production |
| | Habitat characteristics | Extent |
| | Coastal processes | Sediment cycling |
| | Herbivory | Grazing rates |

indicators of MPA ecological conditions include numeric density, biomass, and age and size structure of fish and invertebrates; ecological community-level indicators include species richness and diversity, trophic structure, and functional redundancy; and ecosystem-level attributes include habitat characteristics and key ecological processes (Table 3) (e.g., Refs. 113–115). In general, increases in these metrics suggest positive MPA impacts, although ecological dynamics associated with some "effective" MPAs may result in declines in some indicators (e.g., prey species decline following recovery of predators).[116,117]

MPAs and other conservation interventions directly and indirectly influence human well-being through multiple pathways and mechanisms.[89,118] By introducing new systems of marine resource governance, MPAs may enhance or reshape the flows of marine ecosystem services,[22,119] including provisioning services (e.g., finfish and invertebrate fisheries); regulatory services (e.g., carbon sequestration); cultural services (e.g., tourism); and supporting services (e.g., biodiversity; Table 4).[120–122] MPA governance regimes also define who may access these flows, reallocating the benefits of ecosystem services within and among social groups.[47,123] Thus, MPAs shape the size of the marine resource pie (via ecosystem services) and determine who gets a slice of the pie (via property rights) and the size of the slice allocated to each individual (via property rights). In addition, the infrastructure and ideas associated with a conservation intervention may influence economic activities and social behavior, with direct and indirect consequences for human well-being (e.g., Refs. 124 and 125).

The social impacts (Table 1; O1) of MPAs may be measured using diverse frameworks and indicators (e.g., UN Millennium Development Goals and UN Human Development Index).[126,127] Each framework provides an alternative structure for characterizing human well-being, but they frequently share core constructs, such as economic well-being, health, security and political empowerment, education, and culture.[22,43,128] Within each domain, indicators (Table 5) may be measured at one or more levels of social organization.[129] In Indonesia, for example, indices of material assets measure economic well-being at household,[130] settlement,[131] and district levels.[132] Measuring multiple dimensions of human well-being allows one to examine synergies, trade-offs, and equity among outcomes.[123,133]

MPA impacts have ripple effects in space, time, and among outcomes (often referred to as "spillovers" or "leakage"; these impacts are both dynamic and complex, as studies of fishers have shown).[57,63,134] For example, the redistribution of fishing effort from an MPA can increase the cost of fishing and lower profitability as a result of increased travel distance (and associated costs), increased exploratory fishing, and aggregation of fishing pressure in non-MPA locations.[47,51,135] Observational studies[34,35,86,136] and bioeconomic models[137,138] indicate that stock recovery and ecological spillover (net emigration of adult biomass and larvae) from areas where fishing is

**Table 4. Ecosystem service domains, attributes, and illustrative indicators of the ecosystem service impacts of marine protected areas (MPAs)**[120–122]

| Domain | Attribute | Illustrative indicator |
|---|---|---|
| Provisioning | Fisheries | Fisheries catch |
| | Materials | Volume mined |
| Supporting | Biodiversity | Number of species |
| Regulatory | Carbon sequestration | Mangrove biomass |
| Cultural | Tourism | Annual value of tourism |
| | Recreation | Scuba diving annual value |
| | Existence values | Nonuse existence value |

prohibited can result in increased fish abundance and catch per unit effort in fishable areas, thereby mitigating some or all of these costs. In addition to changes in fishing costs and catch, shifts in the catchability of target species and redistribution of fishing effort result in market feedbacks (e.g., prices, quantity demanded and supplied); fishers may subsequently adapt by changing their harvest strategies.[44,51,57] Bioeconomic models and theory provide insights into these complex interactions and adaptive behavioral strategies, predicting how MPA economic impacts will vary over time and space (e.g., Refs. 63, 137, and 139). These spillovers influence research design, necessitating that studies address the manner in which these ripple effects may influence observations in non-MPA locations. If spillovers result in improved ecological conditions at a control site, for example, this will reduce differences in conditions inside and outside the MPA, resulting in a more conservative estimate of MPA impacts.[24] Establishing control sites at a distance beyond the expected influence of such spillovers allows for more reliable estimates of MPA impacts (see Ref. 70 and the case study described below).

### *Context*

Elucidating the relationship between MPA interactions and outcomes requires controlling for contextual factors that may affect the likelihood of interventions or the nature of impacts. Marine resource governance (Fig. 1; GS), human activities (I), and ecological and social outcomes (O), for example, are shaped by the ecological (ECO) and social, economic, and political (S) context within which they are embedded.[78,140] Context may manifest in a variety of ways, across a variety of scales, influencing the likelihood of a conservation intervention occurring,[18] its governance characteristics,[141] and, ultimately, its social and ecological impacts.[70] Controlling for context—and other exogenous factors—allows one to measure the impact of an MPA by isolating the causal relationship between the intervention (i.e., MPA establishment) and its impacts (i.e., resultant changes in ecological and social conditions).

Biophysical context (ECO) has a substantive effect on marine resource governance, human behavior, and site-level social and ecological conditions. Conservation interventions are preferentially established in areas of high biological importance (e.g., high biodiversity or endemism)[142,143] and low-opportunity costs of conservation (e.g., remote areas),[144,145] with MPAs predominantly occurring in coastal areas[146]—all illustrations of selection bias in the siting of MPAs. Biophysical context also structures historic and current human uses of the marine environment in and around MPAs; for example, upwelling areas of high productivity are often targeted by fishers, while high wave exposure can deter small-scale fishers.[147] Large-scale biophysical factors, such as temperature, productivity, habitat connectivity, and vulnerability to large storms, have shaped present-day ecosystems and continue to influence MPA ecological conditions.[148] Localized biophysical characteristics (e.g., wave exposure, depth, reef type, and benthic habitat characteristics) similarly influence the ecological community structure of MPAs and MPA responses to natural and anthropogenic disturbances—and conservation interventions.[149,150] For example, the ecological recovery of corals within MPAs, which are associated with increases in herbivorous parrotfish, can be greater in sheltered areas where algae grow slowly.[151] These biophysical characteristics may subsequently influence MPA impacts on human well-being, through ecological changes that have cascading effects on social conditions (e.g., ecosystem productivity may influence the intensity, duration, and yield of fishing efforts).

Social context (S) also shapes marine resource governance, human activities, and social and ecological impacts. Social structure influences the probability of collective action and self-governance, with decentralized governance structures more likely to emerge where the probability of local collective action is high.[152] Traditions of customary marine tenure, for example, may foster local MPA

**Table 5. Social domains, attributes, and illustrative indicators of the impacts of marine protected areas**

| Domain | Attribute | Illustrative indicator |
|---|---|---|
| Economic well-being | Occupation | Primary occupation |
| | Economic status | Household material assets index<br>Trend in economic status |
| | Fishing characteristics | Primary fishing technique<br>Household dependence on marine protein<br>Economic dependence on marine fisheries |
| Health | Food security | Household food security index<br>Child food security index<br>Catch per unit effort |
| | Birth rate | Birth rate<br>Adolescent birth rate |
| | Mortality | Infant mortality rate<br>Under-five mortality rate<br>Adult mortality rate |
| | Morbidity | Household disease rate<br>Household disease burden |
| Political empowerment | Resource rights | Household marine tenure index |
| | Community organization | Participation in community groups<br>Financial contribution to marine community groups<br>Female participation in marine community groups |
| | Political engagement | Voting rates |
| Education | Formal education | Adult literacy rates<br>School enrollment rates |
| | Environmental education | Awareness of threats to marine environment<br>Awareness of marine conservation actions |
| Culture | Place attachment | Mean place attachment |
| | Social conflict | Trends in social conflict |

Note: All indicators may be disaggregated by demographic characteristics (e.g., age, gender, wealth, education, and class) to explore the distributive impacts of MPAs.[42,47,118,130]

establishment.[6] At the same time, a lack of human capacity may limit the potential for MPA comanagement or collaborative adaptive management.[153] National social context, by contrast, explains little variation in MPA abundance and spatial extent;[18] for example, fishers per capita does not influence MPA establishment at the national level.[18] Social structures (e.g., occupational structures and resource dependence) also shape MPA use rights and other aspects of marine resource governance.[141,154] Differences in social systems (e.g., societal values and demographic trends (S2)) may also influence MPA impacts; social cohesion, for example, correlates positively with fisheries management outcomes.[155]

Similarly, economic context influences marine resource governance, human activities, and ecological and social conditions. MPAs are often situated where opportunity costs for extractive uses are low,[145] creating considerable selection bias that must be accounted for when comparing MPA to non-MPA locations.[70] Market access (Table 1, S5), for example, may influence MPA establishment and governance.[141] Market access also influences resource use and ecological conditions, as well as human well-being in coastal settings;[140,156,157] economic development has also been linked to fish biomass.[158] Market access may also influence the extent to which local communities depend on marine resources, which, in turn, may mediate the magnitude of MPA impacts on human well-being.[42,63] For example, households with high dependence on marine resources are more likely to be affected by the reallocation of resource rights linked to MPA establishment.[42] More broadly, the economic value of fishery resources has been shown to determine when and where fisheries develop[159] and whether overexploitation is likely to occur.[160] Fishing activity is heavily influenced by the opportunity costs of fishing (i.e., the next best alternative income)[63] and economic incentives, where fishing costs (including distance) and expected returns (price, catch per unit effort) influence fishing locations, intensity, frequency, and gear choice.[44,57]

Lastly, political context (S) may also shape marine resource governance, human activities, and social and ecological conditions.[6,18,69,157] Country-level governance characteristics (e.g., political stability (S3), voice, and accountability) do not appear closely correlated with MPA establishment, though enabling legislation can accelerate MPA establishment (e.g., the Philippines),[18] and shifts in marine resource governance may track the evolution of governance in postcolonial states.[6] Decision-making arrangements may also influence MPA size and location; scattered evidence suggests that decentralization policies foster many small coastal MPAs, whereas centralized decision making is associated with fewer, larger, and more remote MPAs.[18,161] One might expect rule of law, corruption, accountability, and other contextual political factors to shape human behavior within MPAs—and MPA impacts—through macrolevel influences on fishing practices, tourism, or land use, but research has not yet examined these relationships fully (but see Ref. 162). The history of marine resource governance may shape ecological and social context and thus the probability of MPA establishment and/or the nature of MPA impacts, though rigorous research designs should control for such histories and other aspects of political context (for an example, see Box 1).

**Box 1. 3FIGS in action: impact evaluation of MPAs in the Bird's Head Seascape, West Papua, Indonesia**

In the Bird's Head Seascape (BHS) of West Papua, Indonesia, an interdisciplinary team[d] is applying the 3FIGS approach to document and explain the social and ecological impacts of MPAs (Fig. 5).[70,130] The BHS is recognized for its exceptional cultural and biological diversity, but human activities (e.g., destructive fishing practices and coastal sedimentation) have jeopardized both the ecological integrity of the region and the livelihoods of its resource-dependent residents.[163] Given these challenges, local and international organizations partnered to establish 12 MPAs within the BHS, encompassing more than 35,000 km$^2$. Through these MPAs, "empowered and capable Papuan communities, governments, and local partners protect their critical coastal and marine ecosystems, thereby protecting the single greatest reservoir of tropical marine species on the planet, while enhancing food security, livelihood opportunities, as well as their cultural heritage and traditional ownership rights."[164] These MPAs, which include both multiple-use areas and no-fishing areas, establish preferential use rights for local residents that enable these residents to fish for subsistence and commercial use within the MPAs and empower them to exclude nonresidents ineligible to fish within the MPAs.

To determine the social and ecological impacts of the BHS MPAs and to test the null hypothesis of "no impact," the scientific team employed a quasiexperimental impact evaluation informed by SESs to identify both outcome variables and potential confounders. To measure social impacts, the team first established baselines for eight MPAs and matched non-MPA controls. In addition to basic demographic information, household surveys documented human well-being (O1) using more than 150 indicators across five social domains: economic well-being, health, political empowerment, education, and culture. To identify non-MPA control households, the team used a multistep matching process at each MPA, designed to control for confounding processes (identified from the SES classificatory framework). An initial coarse match identified non-MPA settlements with key characteristics similar to MPA settlements. Drawing on the SES framework, coarse matching variables included occupation, social structure, market access, and political jurisdiction as proxies for the key attributes of context (S), resource systems (RS), resource units (RU), and actors (A) that may affect participation in an MPA or its outcomes. After the team surveyed these settlements (having used power analysis to determine the appropriate sample sizes for each settlement and each MPA in its entirety), it used Mahalanobis metric matching (a nearest-neighbor statistical matching method) to identify pairs of MPA and similar non-MPA households with similar observable characteristics. Statistical matching procedures, within an impact evaluation research

[d]The team is led by authors M.B. Mascia, H.E. Fox, L. Glew, and G. Ahmadia, as well as professor Fitryanti Pakiding of the University of Papua in Indonesia.

design, allowed the team to control for observable characteristics related to socioeconomic attributes (A2), ethnic composition (A6), and number of actors (A1) that could influence subgroup treatment effects (e.g., relative impacts on fisher versus nonfisher households). Baseline monitoring occurred in 2010–2012, following legal establishment of the MPAs and designation of boundaries, but before development of management plans and implementation of MPA zoning plans. To minimize the potential for spillover or leakage effects to bias estimates of MPA social impacts, control households and their fishing grounds were identified beyond known distances for economically meaningful levels of fish population spillover. Repeat data collection occurs at 2-year intervals until 4 years postbaseline, at which point monitoring will occur at 3-year intervals. The team uses difference-in-difference estimates of statistically matched MPA and non-MPA households to calculate the average treatment effect (i.e., impact) of each MPA for each of the social indicators. Demographic and spatial data allow the team to explore a targeted subset of spatial and temporal synergies, trade-offs, and equity within and among social groups and across domains of human well-being (O1), as well as to test hypotheses derived from theories of common-pool resource governance on the role of governance in shaping social impacts.

To monitor the ecological impacts of BHS MPAs, the scientific team adopted an analogous research design to measure changes in common reef ecosystem indicators (O2). Indicators included benthic (coral, other invertebrate and algal communities) and fish community attributes (e.g., abundance, size, and biomass of key reef species). After initially establishing ecological baselines within MPA boundaries (2009–2011), the team subsequently employed coarse matching methods to identify corresponding non-MPA controls (on the basis of 10 contextual variables describing structural, biophysical, and social features of coral reef sites (ECO, S))[70] and completed ecological baselines in control sites in 2012. Transect-to-transect statistical matching allows direct "apples-to-apples" comparisons of MPA and non-MPA coral reefs. Subsequent monitoring is occurring at 2-year intervals through baseline +4 years, at which point monitoring will occur at 2- to 3-year intervals. To reduce the risk of contaminated estimates of ecological impacts, control sites were sited beyond estimated distances for ecologically meaningful levels of fish population spillover.[70] As with social impacts, the team uses difference-in-difference estimates of changes in statistically matched MPA and non-MPA underwater transects to calculate the average treatment effect of each MPA across the ecological indicators. Life history and spatial data allow the team to explore a targeted subset of spatial and temporal synergies, trade-offs, and heterogeneities within and among ecological groups and across domains of ecological condition (O2). Larval dispersal and fish migratory behaviors can generate large-scale spillover effects, constraining the effectiveness of conventional use of fixed distance to minimize bias arising from interactions between intervention and nonintervention sites.[70] Consequently, estimates of MPA impact are interpreted with caution, particularly for highly mobile species.

To explain these social and ecological impacts and to test the null hypothesis that governance does not influence MPA outcomes, the team also monitors marine resource governance (decision-making arrangements (GS4), resource-use rights (GS5, GS6), monitoring and enforcement systems (GS8, I9, I10), and conflict-resolution mechanisms) in both MPA settlements and similar non-MPA controls.[130] Linked focus group discussions and key informant interviews provide data on key governance variables, allowing the team to test hypotheses derived from theories of common-pool resource governance. The team employs purposive sampling techniques to identify focus group participants and key informants in each settlement, given that knowledge of governance processes is not distributed uniformly or randomly among residents. Data collection among settlements occurs over two sampling periods; the data are pooled to estimate values for each governance variable at each time step. Governance data allow the team to explore the role of specific governance variables (GS) in shaping the direction, magnitude, and distribution of the social and ecological impacts of MPAs (O).

## Discussion

Understanding what works, what does not, and why outcomes differ are fundamental challenges for conservation science and policy. A longstanding focus on ecological dynamics and human threats to biodiversity has led much of conservation to focus on what to conserve and where to do so.[165,166] A proliferation of initiatives, employed by government agencies and nongovernmental organizations, has emerged in response to these threats.[2] The limited scientific inquiry into the impacts of these initiatives—both to make sense of the world and to inform effective policy—has been plagued by a lack of social theory[91] and by weak research designs that fail to control for alternate explanations of observed outcomes.[11]

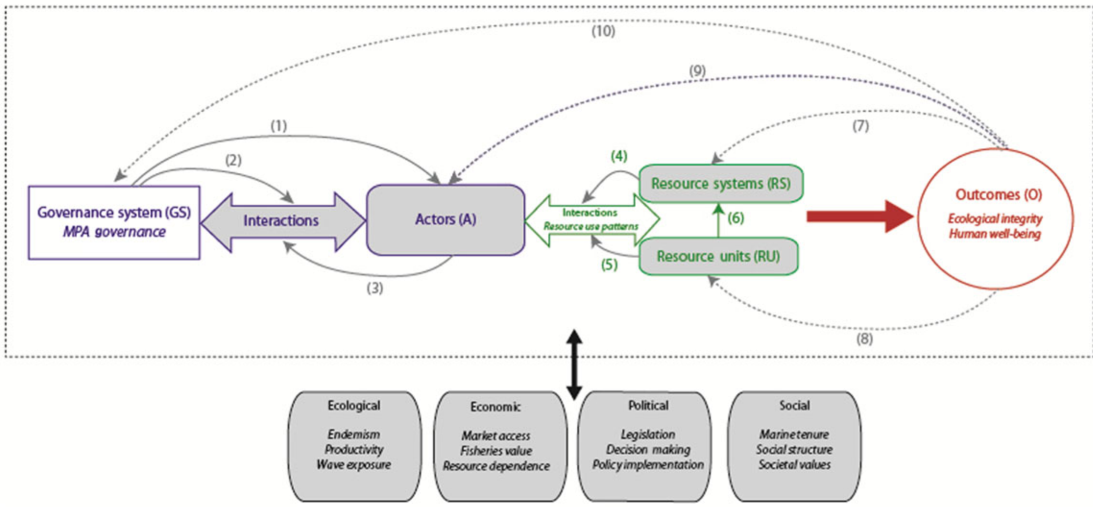


**Figure 5.** Application of the 3FIGS approach to understand the impacts of marine protected areas in the Bird's Head Seascape, Indonesia. Conservation interventions often represent novel governance systems that induce a causal chain of events with intended and unintended consequences, feedback loops, synergies, and trade-offs. Related ecosystems (ECO) and social, economic, and political settings (S) in Figure 1 are collectively represented here as contextual variables. Solid lines represent direct links; dashed lines represent feedback mechanisms. Numbered arrows represent links labeled in Figure 1 as (1) "set rules for," (2) "set conditions for," (3) "participate in," (4) "set conditions for," (5) "are inputs to," and (6) "are part of." Numbered arrows (7–10) represent feedback mechanisms. Gray arrows and filled gray boxes represent elements of the social–ecological system that are explicitly controlled for through quasiexperimental research design.

The social impacts of conservation interventions remain particularly poorly understood.[22,167] As a result, in many cases, conservation policymakers are shooting in the dark, not knowing which interventions work and which do not.[11,168,169]

The 3FIGS approach provides a novel and robust means for documenting and explaining the intended and unintended impacts of MPAs and analogous biodiversity conservation interventions. Impact evaluations of terrestrial and freshwater protected areas, for example, could readily adopt the 3FIGS framework and adapt the MPA example to other biomes. Similarly, 3FIGS could be readily adapted for other place-based conservation interventions that focus on resource governance, such as forest and fisheries certification, payments for ecosystem services, and community-based natural resource management. For conservation interventions that focus on mechanisms other than governance (e.g., environmental education and economic reforms), the theoretical component of 3FIGS can be modified from common-pool resource governance to accommodate other social theories and associated hypotheses (e.g., theories of planned behavior, commodity chains). Appropriate research designs, theories, and conceptual frameworks allow one to measure the direction, magnitude, and distribution of social and ecological impacts that emerge from conservation interventions, as well as the heterogeneity of these impacts among groups and across space and time. Examining MPAs and other conservation interventions as policy experiments provides the basis for both better understanding the world around us and designing more effective conservation interventions.[11,170,171]

### *Caveats*

Despite the power of a 3FIGS approach, however, integrating impact-evaluation research design, the SES conceptual framework, and hypotheses from common-pool resource governance theory is not a silver bullet for understanding the complex synergies and trade-offs that may emerge from MPAs and other conservation interventions. Furthermore, which interventions work and which do not may depend on the context, which is important for the interpretation and transferability to other contexts of impact evaluation results, which have high internal validity but low external validity. Consequently, the practical application of the 3FIGs approach may be easiest where harvested species have limited mobility (minimizing spillover effects) and where an intervention affects a subset of the SES resource (creating a pool of viable control sites). The reductionist framework provided by impact evaluation imposes clear limits, both practical (e.g., complexity of sampling design)[82] and analytic (e.g., subgroup treatment effects),[172,173] on the scope of any given analysis; a reductionist approach may be insufficient to understand the full suite of feedbacks and interactions between social–ecological domains that are known to exist but remain poorly documented.[174] A classic example is “fishing the line,”[175] where an ecological response (i.e., higher fish abundance in the MPA spilling over beyond the boundary) drives a change in human behavior (i.e., spatial reallocation of fishing effort to the MPA boundary), which, in turn, may alter the ecological response (i.e., differential mortality rates alter fish behavior, lowering fish density close to MPA boundaries).[175] Similarly, identifying the unique contributions of factors that may covary with governance is a substantive challenge; for example, administrative activities and capacities, such as management plans, budgets, and staffing, have been hypothesized to shape impacts of MPAs and other conservation interventions,[176] but supporting evidence is weak.[177–181] However, a recent study examining the relationship between management capacity and ecological performance of MPAs has shown that capacity gaps (e.g., budget and staffing) may explain the unrealized conservation potential of MPAs.[24]

While novel analytical approaches (e.g., Ref. 89) increasingly enable researchers to explore complex causal mechanisms and interaction effects, minimum viable sample size[182] and the danger inherent to multiple hypothesis testing[183] impose hard limits on model specification. In effect, the analytic constraints of impact evaluation will require scholars to choose between (1) “black box” approaches that are blind to the governance attributes of the intervention to capture emergent properties and feedbacks between outcomes and (2) carefully bounded analyses that explore the relationships among a subset of governance hypotheses and outcomes. Narrative or modeling approaches, buttressed by specific, testable hypotheses, may be required to disentangle complex interactions.[184]

### Frontiers

Novel data-acquisition tools and analytic techniques may enable scholars to provide insights into increasingly complex questions about the impacts of MPAs. Crowd-sourced data and citizen science, for example, may provide an opportunity to build sufficiently large data sets to understand specific sets of emergent properties and interactions. Recent high-profile MPA research has benefited from widespread observations by nonscientist scuba divers;[38] marine citizen science has also been used to examine global population trends among manta rays.[185] Similarly, the increasing availability of government micro-data in online archives (e.g., the World Bank micro-data archives) will expand the geographic scope of global synthesis studies, including explorations of MPA social impacts. However, the emergence of Big Data alone will not be sufficient to provide empirical insights on 3FIGS. Bayesian statistical techniques that allow for the integration of qualitative data into quantitative models (e.g., as informative priors) and enhanced ability to document the uncertainty inherent to specific estimates may both broaden the suite of hypotheses that can be addressed in a single model and provide more nuanced insights into the emergent properties of SESs.

Some familiar challenges will likely remain, despite novel data-acquisition approaches and analytic advances. Data-synthesis efforts require more data than can be gathered by individual researchers or research projects, particularly for interdisciplinary analysis. Spatial or temporal overlap of ecological and social data-collection efforts is rare, and meaningful integration of multidisciplinary data sets poses both practical and theoretical challenges.[22] MPA synthesis efforts based on published literature provide novel insights[23,42] but are limited by methodological differences in underlying studies (which limit comparability). Synthetic analyses based on raw data are frequently constrained by the ability and willingness of researchers to share data and expertise. Research networks and standardized monitoring approaches provide a framework for data compilation and sharing while also addressing issues of data comparability. Forest- and development-focused research networks, for example, are generating substantive insights on the role of governance in shaping the effectiveness of interventions (e.g., International Forestry Resources and Institutions; Jameel Poverty Action Lab (www.povertyactionlab.org)).[106]

## Conclusions

Despite the challenges associated with impact evaluation of conservation interventions, key scientific and policy advances are possible through a 3FIGS approach. In recent years, independent considerations of impact-evaluation research designs, hypotheses from common-pool resource governance theory, and the SES conceptual frameworks have provided novel insights into the performance of MPAs and other conservation interventions. Scientists now have the opportunity to build upon the strengths of these three approaches to generate fundamental insights into how the world works—the essential basis for evidence-based policy. The conservation community will never answer all of its questions, but greater conceptual clarity and analytic rigor will allow us to better understand how MPAs and other interventions shape society and the world around us.

## Acknowledgments

This work was supported by the National Socio-Environmental Synthesis Center (SESYNC) under funding received from the National Science Foundation DBI-1052875, as part of the working group "Solving the Mystery of Marine Protected Area (MPA) Performance: Linking Governance, Conservation, Ecosystem Services and Human Well Being." David Gill was supported by the Luc Hoffmann Institute and SESYNC. Alfonso Lombana, Gonzalo Cid, Ruth Gates, Anne Henshaw, Hannah Thomas, and Stephen Woodley also provided valuable contributions. Michael Webster and Carly Strasser provided valuable feedback on the manuscript, and Dawn Pointer McCleskey and Molly Spiegel assisted with references. This manuscript is an output of the IUCN World Commission on Protected Areas–Species Survival Commission Joint Task Force on Biodiversity and Protected Areas. This manuscript is contribution #10 of the research initiative *Solving the Mystery of MPA Performance*.

## References

1. Adams, W.M. 2004. *Against Extinction: The History of Conservation.* Earthscan.
2. Rands, M.R.W. *et al.* 2010. Biodiversity conservation: challenges beyond 2010. *Science* **329:** 1298–1303.
3. Adams, W.M. *et al.* 2004. Biodiversity conservation and the eradication of poverty. *Science* **306:** 1146–1149.
4. Gómez-Baggethun, E., R. De Groot, P.L. Lomas & C. Montes. 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. *Ecol. Econ.* **69:** 1209–1218.
5. Dressler, W. *et al.* 2010. From hope to crisis and back again? A critical history of the global CBNRM narrative. *Environ. Conserv.* **37:** 5–15.
6. Johannes, R.E. 2002. The renaissance of community-based marine resource management in Oceania. *Ann. Rev. Ecol. Syst.* **33:** 317–340.
7. Vandenbergh, M.P. 2013. Private environmental governance. *Cornell Law Rev.* **99:** 129–199.
8. Bray, Z. 2010. Reconciling development and natural beauty: the promise and dilemma of conservation easements. *Harvard Environ. Law Rev.* **34:** 119–177.
9. Langholz, J.A. & J.P. Lassoie. 2001. Perils and promise of privately owned protected areas. *Bioscience* **51:** 1079–1085.
10. Costello, C., S.D. Gaines & J. Lynham. 2008. Can catch shares prevent fisheries collapse? *Science* **321:** 1678–1681.
11. Ferraro, P.J. & S.K. Pattanayak. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biol.* **4:** 482–488.
12. Miteva, D.A., S.K. Pattanayak & P.J. Ferraro. 2012. Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxf. Rev. Econ. Pol.* **28:** 69–92.
13. Kelleher, G. & R. Kenchington. 1992. Guidelines for establishing marine protected areas. IUCN, Gland, Switzerland.
14. Juffe-Bignoli, D. *et al.* 2014. Protected planet report 2014. UNEP-WCMC, Cambridge, UK.
15. Thomas, H.L. *et al.* 2014. Evaluating official marine protected area coverage for Aichi Target 11: appraising the data and methods that define our progress. *Aquat. Conserv.* **24:** 8–23.
16. Mascia, M.B. 2004. Social Dimensions of Marine Reserves. In *Marine Reserves: A Guide to Science, Design, and Use.* C. Dahlgren & J. Sobel, Eds.: 164–186. Island Press.
17. Spalding, M.D., I. Meliane, A. Milam, *et al.* 2013. Protecting Marine Spaces: Global Targets and Changing Approaches. In *Ocean Yearbook 27.* Vol. 27. A. Chircop, S. Coffen-Smout & M. McConnell, Eds.: 213–248. Brill Nijhoff.
18. Fox, H.E. *et al.* 2012. Explaining global patterns and trends in marine protected area (MPA) development. *Mar. Policy* **36:** 1131–1138.
19. Jones, P.J.S. 1994. A review and analysis of the objectives of marine nature reserves. *Ocean Coast. Manag.* **24:** 149–178.
20. Jones, P.S. 2001. Marine protected area strategies: issues, divergences and the search for middle ground. *Rev. Fish Biol. Fish.* **11:** 197–216.
21. Toonen, R.J. *et al.* 2013. One size does not fit all: the emerging frontier in large-scale marine conservation. *Marine Pollut Bull.* **77:** 7–10.

22. Fox, H.E. *et al.* 2012. Reexamining the science of marine protected areas: linking knowledge to action. *Conserv. Lett.* **5:** 1–10.
23. Lester, S.E. *et al.* 2009. Biological effects within no-take marine reserves: a global synthesis. *Mar. Ecol. Prog. Ser.* **384:** 33–46.
24. Gill, D.A. *et al.* 2017. Capacity shortfalls hinder the performance of marine protected areas globally. *Nature* **543:** 665–669.
25. Halpern, B.S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? *Ecol. Appl.* **13:** S117–S137.
26. Pinnegar, J.K. *et al.* 2000. Trophic cascades in benthic marine ecosystems: lessons for fisheries and protected-area management. *Environ. Conserv.* **27:** 179–200.
27. Mumby, P.J. *et al.* 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. *Science* **311:** 98–101.
28. Mumby, P.J. & A.R. Harborne. 2010. Marine reserves enhance the recovery of corals on Caribbean reefs. *PLoS One* **5:** 7.
29. Babcock, R.C. *et al.* 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. *Proc. Natl. Acad. Sci. USA* **107:** 18256–18261.
30. Micheli, F. *et al.* 2012. Evidence that marine reserves enhance resilience to climatic impacts. *PLoS One* **7:** e40832.
31. Mumby, P.J., A. Hastings & H.J. Edwards. 2007. Thresholds and the resilience of Caribbean coral reefs. *Nature* **450:** 98–101.
32. McCook, L.J. *et al.* 2010. Adaptive management of the Great Barrier Reef: a globally significant demonstration of the benefits of networks of marine reserves. *Proc. Natl. Acad. Sci. USA* **107:** 18278–18285.
33. Harrison, H.B. *et al.* 2012. Larval export from marine reserves and the recruitment benefit for fish and fisheries. *Curr. Biol.* **22:** 1023–1028.
34. Abesamis, R.A. & G.R. Russ. 2005. Density-dependent spillover from a marine reserve: long-term evidence. *Ecol. Appl.* **15:** 1798–1812.
35. Amargos, F.P. *et al.* 2010. An experiment of fish spillover from a marine reserve in Cuba. *Environ. Biol. Fish.* **87:** 363–372.
36. Januchowski-Hartley, F.A., N.A.J. Graham, J.E. Cinner & G.R. Russ. 2013. Spillover of fish naivete from marine reserves. *Ecol. Lett.* **16:** 191–197.
37. Lester, S.E. & B.S. Halpern. 2008. Biological responses in marine no-take reserves versus partially protected areas. *Mar. Ecol. Prog. Ser.* **367:** 49–56.
38. Edgar, G.J. *et al.* 2014. Global conservation outcomes depend on marine protected areas with five key features. *Nature* **506:** 216–220.
39. Claudet, J. *et al.* 2008. Marine reserves: size and age do matter. *Ecol. Lett.* **11:** 481–489.
40. Claudet, J. *et al.* 2010. Marine reserves: fish life history and ecological traits matter. *Ecol. Appl.* **20:** 830–839.
41. Sciberras, M. *et al.* 2015. Evaluating the relative conservation value of fully and partially protected marine areas. *Fish Fish.* **16:** 58–77.
42. Mascia, M.B., C.A. Claus & R. Naidoo. 2010. Impacts of marine protected areas on fishing communities. *Conserv. Biol.* **24:** 1424–1429.
43. Gurney, G.G. *et al.* 2014. Poverty and protected areas: an evaluation of a marine integrated conservation and development project in Indonesia. *Glob. Environ. Change* **26:** 98–107.
44. Van De Geer, C., M. Mills, V.M. Adams, *et al.* 2013. Impacts of the Moreton Bay Marine Park rezoning on commercial fishermen. *Mar. Policy* **39:** 248–256.
45. Christie, P. 2004. Marine protected areas as biological successes and social failures in Southeast Asia. *Am. Fish. Soc. Symp.* **42:** 155–164.
46. Schlager, E. & E. Ostrom. 1992. Property-rights regimes and natural resources: a conceptual analysis. *Land Econ.* **68:** 249–262.
47. Mascia, M.B. & C.A. Claus. 2009. A property rights approach to understanding human displacement from protected areas: the case of marine protected areas. *Conserv. Biol.* **23:** 16–23.
48. Leisher, C., P. van Beukering & L.M. Scherl. 2007. *Nature's Investment Bank: How Marine Protected Areas Contribute to Poverty Reduction.* Arlington, Virginia: The Nature Conservancy.
49. Bennett, N.J. & P. Dearden. 2014. Why local people do not support conservation: community perceptions of marine protected area livelihood impacts, governance and management in Thailand. *Mar. Policy* **44:** 107–116.
50. Cinner, J.E. *et al.* 2012. Comanagement of coral reef social–ecological systems. *Proc. Natl. Acad. Sci. USA* **109:** 5219–5222.
51. Stevenson, T.C., B.N. Tissot & W.J. Walsh. 2013. Socioeconomic consequences of fishing displacement from marine protected areas in Hawaii. *Biol. Conserv.* **160:** 50–58.
52. Hattam, C., S. Mangi, S. Gall & L. Rodwell. 2014. Social impacts of a temperate fisheries closure: understanding stakeholders' views. *Mar. Policy* **45:** 269–278.
53. Voyer, M., W. Gladstone & H. Goodall. 2014. Understanding marine park opposition: the relationship between social impacts, environmental knowledge and motivation to fish. *Aquat. Conserv.* **24:** 441–462.
54. Gjertsen, H. 2005. Can habitat protection lead to improvements in human well-being? Evidence from marine protected areas in the Philippines. *World Dev.* **33:** 199–217.
55. Myers, S.S. *et al.* 2013. Human health impacts of ecosystem alteration. *Proc. Natl. Acad. Sci. USA* **110:** 18753–18760.
56. Sumaila, U.R. & A.T. Charles. 2002. Economic models of marine protected areas: an introduction. *Nat. Resour. Model.* **15:** 261–272.
57. Smith, M.D. & J.E. Wilen. 2003. Economic impacts of marine reserves: the importance of spatial behavior. *J. Environ. Econ. Manage.* **46:** 183–206.
58. Charles, A. & L. Wilson. 2009. Human dimensions of marine protected areas. *ICES J. Mar. Sci.* **66:** 6–15.
59. Charles, A. 2010. Fisheries and marine protected areas: a spatial bioeconomic analysis of distributional impacts. *Nat. Resour. Model.* **23:** 218–252.
60. Hargreaves-Allen, V., S. Mourato & E.J. Milner-Gulland. 2011. A global evaluation of coral reef management performance: are MPAs producing conservation and socioeconomic improvements? *Environ. Manage.* **47:** 684–700.

61. Ransom, K.P. & S.C. Mangi. 2010. Valuing recreational benefits of coral reefs: the case of Mombasa Marine National Park and Reserve, Kenya. *Environ. Manage.* **45:** 145–154.
62. Terk, E. & N. Knowlton. 2010. The role of SCUBA diver user fees as a source of sustainable funding for coral reef marine protected areas. *Biodiversity* **11:** 78–84.
63. Smith, M.D., J. Lynham, J.N. Sanchirico & J.A. Wilson. 2010. Political economy of marine reserves: understanding the role of opportunity costs. *Proc. Natl. Acad. Sci. USA* **107:** 18300–18305.
64. Pomeroy, R.S., M.B. Mascia & R.B. Pollnac. 2007. Marine Protected Areas: The Social Dimension. In *Report and documentation of the expert workshop on marine protected areas and fisheries management: review of issues and considerations.* pp. 149–182. FAO, Rome. FAO Fisheries Report No. 825.
65. Ward, T.J., D. Heinemann & N. Evans. 2001. The role of marine reserves as fisheries management tools. *A review of concepts, evidence and international experience.* Bureau of Rural Sciences, Canberra.
66. Pomeroy, R.S., J.E. Parks & L.M. Watson. 2004. How is your MPA doing? A guide book of natural and social indicators for evaluating marine protected area management effectiveness. IUCN, Gland, Switzerland and Cambridge, UK.
67. CMP (Conservation Measures Partnership). 2013. Open standards for the practice of conservation. Version 3.0. Washington, DC: Conservation Measures Partnership.
68. Salafsky, N. & R. Margoluis. 1998. *Measures of Success: Designing, Managing, and Monitoring Conservation and Development Projects.* Island Press.
69. Pollnac, R. *et al.* 2010. Marine reserves as linked social–ecological systems. *Proc. Natl. Acad. Sci. USA* **107:** 18262–18265.
70. Ahmadia, G.N. *et al.* 2015. Integrating impact evaluation in the design and implementation of monitoring marine protected areas. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **370:** 20140275.
71. Gruby, R.L., N.J. Gray, L.M. Campbell & L. Acton. 2016. Toward a social science research agenda for large marine protected areas. *Conserv. Lett.* **9:** 153–163.
72. Folke, C. *et al.* 2010. Resilience thinking: integrating resilience, adaptability and transformability. *Ecol. Soc.* **15:** 20.
73. Gertler, P.J., S. Martinez, P. Premand, *et al.* 2011. *Impact Evaluation in Practice.* Washington, DC: The World Bank.
74. McGinnis, M.D. & E. Ostrom. 2014. Social–ecological system framework: initial changes and continuing challenges. *Ecol. Soc.* **19:** 30.
75. Berkes, F., C. Folke & J. Colding. 2000. *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience.* Cambridge University Press.
76. Folke, C. 2006. Resilience: the emergence of a perspective for social–ecological systems analyses. *Glob. Environ. Change* **16:** 253–267.
77. Ostrom, E. 2007. A diagnostic approach for going beyond panaceas. *Proc. Natl. Acad. Sci. USA* **104:** 15181–15187.
78. Ostrom, E. 2009. A general framework for analyzing sustainability of social–ecological systems. *Science* **325:** 419–422.
79. Basurto, X., S. Gelcich & E. Ostrom. 2013. The social ecological system framework as a knowledge classificatory system for benthic small-scale fisheries. *Glob. Environ. Change* **23:** 1366–1380.
80. Basurto, X. & E. Ostrom. 2009. Beyond the tragedy of the commons. *Economia delle fonti di energia e dell' ambiente* **52:** 35–60.
81. Mascia, M.B. *et al.* 2003. Conservation and the social sciences. *Conserv. Biol.* **17:** 649–650.
82. Rosenbaum, P.R. 2010. *Design of Observational Studies.* Springer.
83. GAO (Government Accountability Office). 2009. Program evaluation: a variety of rigorous methods can help identify effective interventions. Washington, DC: U.S. Government Accountability Office.
84. Stuart, E.A. & D.B. Rubin. 2007. Best practices in ANCOVA may mean not using ANCOVA: why paired subjects designs are a better choice. In *Best Practices in Quantitative Social Science.* J. Osborne, Ed.: 155–176. Los Angeles: Sage Publications.
85. Sekhon, J.S. 2009. Opiates for the matches: matching methods for causal inference. *Ann. Rev. Pol. Sci.* **12:** 487–508.
86. da Silva, I.M., N. Hill, H. Shimadzu, *et al.* 2015. Spillover effects of a community-managed marine reserve. *PLoS One* **10:** e0111774.
87. Andam, K.S., P.J. Ferraro, A. Pfaff, *et al.* 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *Proc. Natl. Acad. Sci. USA* **105:** 16089–16094.
88. Joppa, L. & A. Pfaff. 2010. Reassessing the forest impacts of protection: the challenge of nonrandom location and a corrective method. *Ann. N.Y. Acad. Sci.* **1185:** 135–149.
89. Ferraro, P.J. & M.M. Hanauer. 2014. Quantifying causal mechanisms to determine how protected areas affect poverty through changes in ecosystem services and infrastructure. *Proc. Natl. Acad. Sci. USA* **111:** 4332–4337.
90. Mascia, M.B. *et al.* 2014. Commonalities and complementarities among approaches to conservation monitoring and evaluation. *Biol. Conserv.* **169:** 258–267.
91. Agrawal, A. 2014. Matching and mechanisms in protected area and poverty alleviation research. *Proc. Natl. Acad. Sci. USA* **111:** 3909–3910.
92. Ostrom, E. 1990. *Governing the Commons: The Evolution of Institutions for Collective Action.* Cambridge University Press.
93. Ostrom, E. 1998. Self-Governance of Common Pool Resources. In *The New Palgrave Dictionary of Economics and the Law.* P. Newman, Ed. London: Macmillan Reference.
94. Agrawal, A. 2003. Sustainable governance of common pool resources: context, methods, and politics. *Ann. Rev. Anthropol.* **32:** 243–262.
95. Ostrom, E., R. Gardner & J. Walker. 1994. *Rules, Games, and Common Pool Resources.* University of Michigan Press.
96. Ostrom, V. & E. Ostrom. 1977. *Alternatives for Delivering Public Services: Toward Improved Performance.* E. Savas, Ed.: 7–49. Westview Press.

97. Agrawal, A. 2001. Common property institutions and sustainable governance of resources. *World Dev.* **29:** 1649–1672.
98. McCay, B.J. & J.M. Acheson, Eds. 1987. *The Question of the Commons: The Culture and Ecology of Communal Resources.* Tuscon: University of Arizona Press.
99. National Research Council. 1986. *Proceedings of the Conference on Common Property Resource Management.* Washington, DC: National Academy Press.
100. Claudet, J. & P. Guidetti. 2010. Improving assessments of marine protected areas. *Aquat. Conserv.* **20:** 239–242.
101. McClanahan, T., H. Glaesel, J. Rubens & R. Kiambo. 1997. The effects of traditional fisheries management on fisheries yields and the coral-reef ecosystems of southern Kenya. *Environ. Conserv.* **24:** 105–120.
102. Plummer, R., D.R. Armitage & R.C. De Loë. 2013. Adaptive comanagement and its relationship to environmental governance. *Ecol. Soc.* **18:** 21.
103. The World Bank. *What is Governance?* Accessed June 30, 2017. http://web.worldbank.org/WBSITE/EXTERNAL/COUNTRIES/MENAEXT/EXTMNAREGTOPGOVERNANCE/0,contentMDK:20513159∼pagePK:34004173∼piPK:34003707∼theSitePK:497024,00.html.
104. UNDP (United Nations Development Programme), UNEP (United Nations Environment Programme), The World Bank & World Resources Institute. 2003. World resources 2002–2004: decisions for the earth: balance, voice, and power. World Resources Institute.
105. Ostrom, E. 2005. *Understanding Institutional Diversity.* Princeton University Press.
106. Persha, L., A. Agrawal & A. Chhatre. 2011. Social and ecological synergy: local rulemaking, forest livelihoods, and biodiversity conservation. *Science* **331:** 1606–1608.
107. Chandler, W.J. & H. Gillelan. 2004. The history and evolution of the National Marine Sanctuaries Act. *Environ. Law Rep.* **34:** 10505–10562.
108. Obura, D.O., S. Wells, J. Church & C. Horrill. 2002. Monitoring of fish and fish catches by local fishermen in Kenya and Tanzania. *Mar. Freshwater Res.* **53:** 215–222.
109. Bunce, L., P. Townsley, R. Pomeroy & R. Pollnac. 2000. *Socioeconomic Manual for Coral Reef Management.* Australian Institute of Marine Science.
110. Thurstan, R.H., J.P. Hawkins, L. Neves & C.M. Roberts. 2012. Are marine reserves and non-consumptive activities compatible? A global analysis of marine reserve regulations. *Mar. Policy* **36:** 1096–1104.
111. Pinsky, M.L., O.P. Jensen, D. Ricard & S.R. Palumbi. 2011. Unexpected patterns of fisheries collapse in the world's oceans. *Proc. Natl. Acad. Sci. USA* **108:** 8317–8322.
112. Schlager, E., W. Blomquist & S.Y. Tang. 1994. Mobile flows, storage, and self-organized institutions for governing common pool resources. *Land Econ.* **70:** 294–317.
113. Gaston, K.J., S.E. Jackson, L. Cantu-Salazar & G. Cruz-Pinon. 2008. The ecological performance of protected areas. *Ann. Rev. Ecol. Evol. Syst.* **39:** 93–113.
114. Russ, G.R. 2002. *Coral Reef Fishes: Dynamics and Diversity in a Complex Ecosystem.* P.F. Sale, Ed.: 549. Academic Press.
115. Bertness, M.D., S.D. Gaines & M. Hay, Eds. 2001. *Marine Community Ecology* Sunderland, Massachusetts: Sinauer Associates, p. 550.
116. Micheli, F., B.S. Halpern, L.W. Botsford & R.R. Warner. 2004. Trajectories and correlates of community change in no-take marine reserves. *Ecol. Appl.* **14:** 1709–1723.
117. Harborne, A.R., P.G. Renaud, E.H.M. Tyler & P.J. Mumby. 2009. Reduced density of the herbivorous urchin *Diadema antillarum* inside a Caribbean marine reserve linked to increased predation pressure by fishes. *Coral Reefs* **28:** 783–791.
118. Stephanson, S.L. & M.B. Mascia. 2014. Putting people on the map through an approach to integrating social data in conservation planning. *Conserv. Biol.* **28:** 1236–1248.
119. Lubchenco, J., S.R. Palumbi, S.D. Gaines & S. Andelman. 2003. Plugging a hole in the ocean: the emerging science of marine reserves. *Ecol. Appl.* **13:** S3-S7.
120. Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-being: Synthesis.* Island Press.
121. Moberg, F. & C. Folke. 1999. Ecological goods and services of coral reef ecosystems. *Ecol. Econ.* **29:** 215–233.
122. Schuhmann, P.W. & R. Mahon. 2014. The valuation of marine ecosystem goods and services in the Caribbean: a literature review and framework for future valuation efforts. *Ecosyst. Serv.* **11:** 56–66.
123. Daw, T., K. Brown, S. Rosendo & R. Pomeroy. 2011. Applying the ecosystem services concept to poverty alleviation: the need to disaggregate human well-being. *Environ. Conserv.* **38:** 370–379.
124. Ferraro, P.J. & M.M. Hanauer. 2015. Through what mechanisms do protected areas affect environmental and social outcomes? *Philos. Trans. R. Soc. B Biol. Sci.* **370:** 20140267.
125. Robalino, J. & L. Villalobos. 2015. Protected areas and economic welfare: an impact evaluation of national parks on local workers' wages in Costa Rica. *Environ. Dev. Econ.* **20:** 283–310.
126. UN Millennium Project. 2005. Investing in development: a practical plan to achieve the millennium development goals. New York: UNDP. Accessed June 22, 2017. http://www.unmillenniumproject.org/documents/MainReportComplete-lowres.pdf.
127. UNDP (United Nations Development Programme). 1990. Human development report 1990. New York, NY: Oxford University Press. Accessed June 22, 2017. http://hdr.undp.org/sites/default/files/reports/219/hdr_1990_en_complete_nostats.pdf.
128. Woodhouse, E. *et al.* 2015. Guiding principles for evaluating the impacts of conservation interventions on human well-being. *Philos. Trans. R. Soc. Lond. B: Biol. Sci.* **370**. https://doi.org/10.1098/rstb.2015.0103.
129. Bernard, H.R. 2011. *Research Methods in Anthropology: Qualitative and Quantitative Approaches.* AltaMira Press.
130. Glew, L., M.B. Mascia & F. Pakiding. 2012. Solving the mystery of MPA performance: monitoring social impacts. Field manual (version 1.0). World Wildlife Fund and Universitas Negeri Papua, Washington, DC and Manokwari, Indonesia.
131. Badan Pusat Statistik. 2011. *Potensi desa daerah Indonesia.* Badan Pusat Statistik, Jakarta, Indonesia.
132. Badan Pusat Statistik Kabupaten Alor. 2013. *Alor Dalan Angka.* Badan Pusat Statistik, Kalabahi, Indonesia.

133. Agrawal, A. & A. Chhatre. 2011. Against mono-consequentialism: multiple outcomes and their drivers in social–ecological systems. *Glob. Environ. Change* **21:** 1–3.
134. Halpern, B.S. 2004. Confounding effects of the export of production and the displacement of fishing effort from marine reserves. *Ecol. Appl.* **14:** 1248–1256.
135. Chollett, I., S.J. Box & P.J. Mumby. 2016. Quantifying the squeezing or stretching of fisheries as they adapt to displacement by marine reserves. *Conserv. Biol.* **30:** 166–175.
136. Guidetti, P. & J. Claudet. 2010. Comanagement practices enhance fisheries in marine protected areas. *Conserv. Biol.* **24:** 312–318.
137. Sanchirico, J.N. & J.E. Wilen. 2002. The impacts of marine reserves on limited-entry fisheries. *Nat. Resour. Model.* **15:** 291–310.
138. Krueck, N.C. *et al.* 2017. Marine reserve targets to sustain and rebuild unregulated fisheries. *PLoS Biol.* **15:** e2000537.
139. Smith, R.J., R.D.J. Muir, M.J. Walpole, *et al.* 2003. Governance and the loss of biodiversity. *Nature* **426:** 67–70.
140. Brewer, T.D., J.E. Cinner, R. Fisher, *et al.* 2012. Market access, population density, and socioeconomic development explain diversity and functional group biomass of coral reef fish assemblages. *Glob. Environ. Change* **22:** 399–406.
141. Cinner, J.E. 2005. Socioeconomic factors influencing customary marine tenure in the Indo-Pacific. *Ecol. Soc.* **10:** 36.
142. Loucks, C., T.H. Ricketts, R. Naidoo, *et al.* 2008. Explaining the global pattern of protected area coverage: relative importance of vertebrate biodiversity, human activities and agricultural suitability. *J. Biogeogr.* **35:** 1337–1348.
143. Roberts, C.M. *et al.* 2003. Ecological criteria for evaluating candidate sites for marine reserves. *Ecol. Appl.* **13:** S199–S214.
144. Joppa, L.N. & A. Pfaff. 2009. High and far: biases in the location of protected areas. *PLoS One* **4:** 6.
145. Devillers, R. *et al.* 2014. Reinventing residual reserves in the sea: are we favouring ease of establishment over need for protection? *Aquat. Conserv.* **25:** 480–504
146. Wood, L.J., L. Fish, J. Laughren & D. Pauly. 2008. Assessing progress towards global marine protection targets: shortfalls in information and action. *Oryx* **42:** 340–351.
147. McClanahan, T.R. & N.A. Muthiga. 1998. An ecological shift in a remote coral atoll of Belize over 25 years. *Environ. Conserv.* **25:** 122–130.
148. Menge, B.A. & A.M. Olson. 1990. Role of scale and environmental factors in regulation of community structure. *Trends Ecol. Evol.* **5:** 52–57.
149. Knowlton, N. & J.B.C. Jackson. 2008. Shifting baselines, local impacts, and global change on coral reefs. *PLoS Biol.* **6:** 215–220.
150. Bejarano, S., P.J. Mumby & I. Sotheran. 2011. Predicting structural complexity of reefs and fish abundance using acoustic remote sensing (RoxAnn). *Mar. Biol.* **158:** 489–504.
151. Mumby, P.J., N.H. Wolff, Y.M. Bozec, *et al.* 2014. Operationalizing the resilience of coral reefs in an era of climate change. *Conserv. Lett.* **7:** 176–187.
152. Taylor, M. 1987. *The Possibility of Cooperation*. Cambridge: Cambridge University Press.
153. McConney, P. & M. Pena. 2012. Capacity for (co)management of marine protected areas in the Caribbean. *Coast. Manage.* **40:** 268–278.
154. Cinner, J.E. 2007. Designing marine reserves to reflect local socioeconomic conditions: lessons from long-enduring customary management systems. *Coral Reefs* **26:** 1035–1045.
155. Gutierrez, N.L., R. Hilborn & O. Defeo. 2011. Leadership, social capital and incentives promote successful fisheries. *Nature* **470:** 386–389.
156. Cinner, J.E., N.A.J. Graham, C. Huchery & M.A. Macneil. 2013. Global effects of local human population density and distance to markets on the condition of coral reef fisheries. *Conserv. Biol.* **27:** 453–458.
157. Fisher, B., A.M. Ellis, D.K. Adams, *et al.* 2015. Health, wealth, and education: the socioeconomic backdrop for marine conservation in the developing world. *Mar. Ecol. Prog. Ser.* **530:** 233–242.
158. Cinner, J.E. *et al.* 2009. Linking social and ecological systems to sustain coral reef fisheries. *Curr. Biol.* **19:** 206–212.
159. Sethi, S.A., T.A. Branch & R. Watson. 2010. Global fishery development patterns are driven by profit but not trophic level. *Proc. Natl. Acad. Sci. USA* **107:** 12163–12167.
160. Collette, B.B. *et al.* 2011. High value and long life—double jeopardy for tunas and billfishes. *Science* **333:** 291–292.
161. White, A.T. *et al.* 2014. Marine protected areas in the coral triangle: progress, issues, and options. *Coast. Manage.* **42:** 87–106.
162. Arias, A., R.L. Pressey, R.E. Jones, *et al.* 2016. Optimizing enforcement and compliance in offshore marine protected areas: a case study from Cocos Island, Costa Rica. *Oryx* **50:** 18–26.
163. Mangubhai, S. *et al.* 2012. Papuan Bird's Head Seascape: emerging threats and challenges in the global center of marine biodiversity. *Marine Pollut Bull.* **64:** 2279–2295.
164. Katz, L. *et al.* 2015. Blue Abadi business plan. Bird's Head Seascape coalition. Manokwari, West Papua, Indonesia.
165. Brooks, T.M. *et al.* 2006. Global biodiversity conservation priorities. *Science* **313:** 58–61.
166. Margules, C.R. & R.L. Pressey. 2000. Systematic conservation planning. *Nature* **405:** 243–253.
167. Pullin, A.S. *et al.* 2013. Human well-being impacts of terrestrial protected areas. *Environ. Evid.* **2:** 19.
168. Agrawal, A. & K. Redford. 2006. *Poverty, Development and Biodiversity Conservation: Shooting in the Dark*?. Bronx New York: Wildlife Conservation Society.
169. Sutherland, W.J., A.S. Pullin, P.M. Dolman & T.M. Knight. 2004. The need for evidence-based conservation. *Trends Ecol. Evol.* **19:** 305–308.
170. Jensen, O.P., T.A. Branch & R. Hilborn. 2012. Marine fisheries as ecological experiments. *Theor. Ecol.* **5:** 3–22.
171. Campbell, D.T. 1969. Reforms as experiments. *Am. Psychol.* **24:** 409–429.
172. Bloom, H. & C. Michalopoulos. 2013. When is the story in the subgroups? *Prev. Sci.* **14:** 179–188.

173. Schochet, P.Z., M. Puma & J. Deke. 2014. Understanding variation in treatment effects in education impact evaluations: an overview of quantitative methods. U.S. Department of Education, Washington, DC. Report No. NCEE 2014-4017.
174. Rowe, A. 2014. Assessing progress towards impacts in environmental programmes using the field review of outcomes to impacts methodology. In *Evaluating Environment in International Development*. J.I. Uitto, Ed.: 45–61. Routledge.
175. Kellner, J.B., I. Tetreault, S.D. Gaines & R.M. Nisbet. 2007. Fishing the line near marine reserves in single and multispecies fisheries. *Ecol. Appl.* **17:** 1039–1054.
176. Hockings, M., S. Stolton, F. Leverington, *et al.* 2006. *Evaluating Effectiveness: A Framework for Assessing Management Effectiveness of Protected Areas.* 2 ed. IUCN.
177. Nolte, C. & A. Agrawal. 2013. Linking management effectiveness indicators to observed effects of protected areas on fire occurrence in the Amazon rainforest. *Conserv. Biol.* **27:** 155–165.
178. Nolte, C., A. Agrawal & P. Barreto. 2013. Setting priorities to avoid deforestation in Amazon protected areas: are we choosing the right indicators? *Environ. Res. Lett.* **8:** 015039.
179. Carranza, T., A. Manica, V. Kapos & A. Balmford. 2014. Mismatches between conservation outcomes and management evaluation in protected areas: a case study in the Brazilian Cerrado. *Biol. Conserv.* **173:** 10–16.
180. Henschel, P. *et al.* 2014. The lion in West Africa is critically endangered. *PLoS One* **9:** e83500.
181. Maypa, A.P., T.W. Alan, E. Caňares, *et al.* 2012. Marine protected area management effectiveness: progress and lessons in the Philippines. *Coast. Manage.* **40:** 510–524.
182. Quintiles Outcome. 2013. *Developing a Protocol for Observational Comparative Effectiveness Research: A User's Guide.* Publication No. 12(13)-EHC099. P. Velentgas *et al.*, Eds. Rockville, MD: U.S. Agency for Healthcare Research and Quality.
183. Schochet, P.Z. 2009. The estimation of average treatment effects for clustered RCTs of education interventions. U.S. Department of Education, Washington, DC. Report No. NCEE 2009-0061 rev.
184. Bates, R.H., A. Greif, M. Levi, *et al.* 1998. *Analytic Narratives.* Princeton, NJ: Princeton University Press.
185. Ward-Paige, C.A., B. Davis & B. Worm. 2013. Global population trends and human use patterns of Manta and Mobula rays. *PLoS One* **8:** 9.