A comparison of approaches used for economic analysis in marine protected area network planning in California

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ABSTRACT

In addition to fostering biodiversity goals, marine protected area (MPA) implementation has economic consequences for both commercial and recreational fisheries. During the implementation of the State of California (USA) Marine Life Protection Act (MLPA), which mandates the creation of an MPA network in California’s state waters, the stakeholders and policymakers utilized a pair of economic analyses that addressed these considerations. One was a comparative, static assessment of short-term, “worst case” potential socioeconomic impacts to important fisheries based on surveys of local fishermen. This analysis made no assumptions about fishery management outside of MPAs, assumed no spillover of fish from MPAs into fished areas or reallocation of fishing effort, and estimated the maximum potential dollar-value economic impacts over a short time scale. The other was a dynamic, bioeconomic assessment of the changes in spatial distribution of biomass and catch, based on published biological parameter values, oceanographic models of larval connectivity, and a range of possible levels of fishing. This analysis explicitly accounted for fish population dynamics, spillover, fisher movement, and fishery management outside of the MPAs, but was limited to long-term, equilibrium-based results because of a lack of baseline abundance data. Both evaluation methods were novel in their spatial resolution and their use directly in an MPA design process, rather than after implementation. The two methods produced broadly similar (at worst case)” results. The MLPA initiative process led to several suggestions for future MPA design efforts: (i) since the change in fish biomass inside MPAs partly depends on fishing from a location is often associated with habitat recovery and increases in species diversity and the density, biomass, and average size of organisms (Halpern and Warner, 2002; Halpern, 2003; Lester et al., 2009).

1. Introduction

The implementation of marine protected areas (MPAs) has, by necessity, both human and ecological consequences. Restricting or excluding extractive activities such as fishing from a location is often associated with habitat recovery and increases in species diversity and the density, biomass, and average size of organisms (Halpern and Warner, 2002; Halpern, 2003; Lester et al., 2009).
Since fishing is also an important economic activity in many coastal areas, such limitations on fishing effort result in both short and long term impacts on the fishing economy. While there is a growing literature on the economic costs and benefits of MPAs (see review in Sanchirico et al., 2002), MPAs are typically established in anticipation of biological or ecological benefits without explicit consideration of economic factors (Sumaila et al., 2000). Economic effects are typically evaluated only post hoc for MPAs that were designed to meet biological and ecological objectives (Scholz et al., 2004; Stewart and Possingham, 2005).

The economic impacts of MPAs will depend on their spatial configuration — some places will accumulate biomass more effectively and some places will be more costly if excluded as fishing grounds (Costello et al., 2010; Smith and Wilen, 2003; White et al., 2010a). Therefore, effective MPA design requires understanding both the current spatial pattern of fishing in the ocean and how those patterns change in response to MPA implementation. Current fishery management planning practices, however, tend to be nonspatial. For example, the guidelines for economic analysis for federal fisheries management recognize that “combining biological information with fishery economics” is needed for both qualitative and quantitative analysis of fishery management actions (Office of Sustainable Fisheries, 2000, p. 4)”, but then remain silent on the spatial dimension of both biological and economic systems. Additionally, the data routinely used in fisheries management are inadequate for spatial analysis. For example, fishery assessments commonly assume that populations are well-mixed at a scale of several hundreds of km along a coastline. Similarly, fishery catches are not commonly reported at a fine-enough spatial scale and require considerable transformation before they can be used in a spatial context (Scholz et al., 2005). Even then they are not suitable to resolve potential economic impacts of individual MPAs.

Because of the inherently spatial nature of MPAs, models for assessing the impacts of MPAs on fished populations have developed at finer scales of resolution than typical fishery stock assessment models. To date, the state of spatial population modeling efforts for MPAs has been predominantly “strategic”, simply predicting the range of sizes and/or numbers of MPAs and the spacing necessary for population persistence under different fishing regimes along an idealized coastline (e.g., Botsford et al., 2001; Gaylord et al., 2005; Mangel, 1998; Sladek Nowlis and Roberts, 1999; White et al., 2008). There have been few examples of site-specific MPA population models (e.g., Little et al., 2007; Stockhausen et al., 2000; see Pelletier and Mahevas, 2005 for a review of MPA models), primarily because of limited information on larval dispersal.

From 2004 to 2011, the state of California, USA, underwent a public MPA design process resulting in the designation of a statewide network of no-take and restricted-take MPAs. This process was mandated by the 1999 passage of the Marine Life Protection Act (MLPA) in the California legislature (see Kirlin et al., in this issue, for details; see Table 1 for a list of acronyms used in this paper). The MLPA was implemented through a series of regional planning efforts.

In this paper we describe and synthesize two complementary approaches to including economic and population dynamic factors in the evaluation of proposed MPA networks. We developed and applied these two analytical approaches in our roles as scientific advisors to the regional MLPA planning processes:

i) a comparative static assessment of potential maximum short-term economic impacts to regionally-important fisheries that is agnostic of external fishery management and assumes no spillover of fish from reserves to fished areas or reallocation of fishing effort, but does express fishing effort, real local catches and economic values (henceforth the “static” analysis).

ii) dynamic bioeconomic modeling of long-term equilibrium outcomes that explicitly accounts for fish population dynamics, spillover, fisher movement, and fishery management outside of the MPAs (henceforth the “dynamic” analysis).

Importantly, both approaches explicitly consider the spatial nature of fishing activities. Both this aspect and the timing of the economic analysis in the regional planning processes constitute a significant departure from past MPA planning practices worldwide. In this paper, we discuss the advantages and shortcomings of these approaches, lessons learned from the California MPA network planning experience, and next steps for MPA planning in California and elsewhere. This paper is novel in its synthesis of two approaches to economic analysis and the presentation of lessons learned.

2. Methods

2.1. The MLPA process

The details of the enabling legislation and structure of the MLPA planning process are described in detail elsewhere in this volume (Fox et al., in this issue-a, in this issue-b; Kirlin et al., in this issue), but to summarize briefly, the design process proceeded in stages, each stage corresponding to one study region of the California coast (Central, North Central, South, and North, in chronological order). The design process for each region took 1–2 years and involved a Regional Stakeholder Group (RSG) who proposed a suite of alternative MPA configurations, a Science Advisory Team (SAT), who promulgated various ecological guidelines for MPA design and then evaluated how well each proposal met those guidelines (Saarman et al., in this issue), and a Blue Ribbon Task Force (BRTF) composed of policymakers who advised the process and forwarded stakeholder proposals and recommended a preferred alternative proposal for final evaluation and decision by the California Fish and Game Commission. The process was iterative, with the RSG revising proposals over the course of several rounds based on SAT evaluations and BRTF guidance. The MLPA Initiative was a public-private partnership created to oversee and facilitate the process.

<table>
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<tr>
<th>Table 1</th>
<th>Acronyms and abbreviations used in this paper.</th>
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<tr>
<td>Acronym</td>
<td>Definition</td>
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<td>General terms</td>
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<tr>
<td>CPFV</td>
<td>Commercial passenger fishing vessel (charter fishing boat)</td>
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<tr>
<td>GIS</td>
<td>Geographic Information System</td>
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<tr>
<td>MARXAN</td>
<td>Marine Spatially Explicit Annealing (software package)</td>
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<td>MPA</td>
<td>Marine Protected Area</td>
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<td>MSY</td>
<td>Maximum Sustainable Yield</td>
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<td>MLPA process</td>
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<td>BRTF</td>
<td>Blue Ribbon Task Force</td>
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<td>MLPA</td>
<td>Marine Life Protection Act</td>
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<td>RSG</td>
<td>Regional Stakeholder Group</td>
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<td>SAT</td>
<td>Science Advisory Team</td>
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<td>MLPA study regions</td>
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<tr>
<td>CCSR</td>
<td>Central Coast Study Region (Pt. Conception to Pigeon Pt.)</td>
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<tr>
<td>SCSR</td>
<td>South Coast Study Region (Mexican border to Pt. Conception)</td>
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<tr>
<td>NCCSR</td>
<td>North Central Coast Study Region (Pigeon Pt. to Pt. Arena)</td>
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<tr>
<td>NCSR</td>
<td>North Coast Study Region (Pt. Arena to Oregon border)</td>
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Notes: Listed in chronological order; the definitions give each region’s approximate southern and northern boundaries.
2.2. Comparative static analysis

The comparative static analysis evolved over the course of the MLPA process. It originated in the first study region (the Central Coast Study Region, or CCSR), when stakeholders and policymakers requested explicit consideration of the impacts of particular MPA designs on fisheries. While the Act did not contain explicit socioeconomic objectives, the MLPA Initiative responded by commissioning the collection of data on the spatial extent and relative importance of the commercial fishing grounds off the coast of the study region. The methodology built on an approach described in Scholz et al. (2004). Initially applied to commercial fisheries, the data collection and analysis was eventually refined and expanded to include recreational fisheries as well (Scholz et al., 2011).

In each study region, a stratified sample of participants in economically important commercial and, starting with the second study region (the North Central Coast Study Region, or NCCSR), recreational fisheries was surveyed as to the extent of their fishing grounds. These grounds were weighted by relative stated importance, and aggregated across respondents and fisheries. In addition, data on a number of demographic and operating cost variables were collected. The approach was refined in subsequent study regions to administer the surveys earlier and make data products available earlier to the RSG in the planning phase. In addition, it was expanded to include recreational fisheries ranging from charter operators to private anglers, resulting in an increasingly more resolved characterization of the location and economic importance of fishing effort off the coast of California (Fig. 1). Ultimately, in the four study regions of the MLPA, a total of 756 commercial fishermen and operators of 162 commercial passenger fishing vessels (CPFV; the term for multi-passenger charter recreational fishing vessels) were interviewed in person, using a custom GIS application, and another 1158 private recreational anglers participated via an on-line version of the surveys (Scholz et al., 2006, 2008, 2010, 2011).

The static analysis was used to provide four main types of results. First each MPA alternative was described in terms of the areal percentage of fishing grounds that were closed and their value to fisheries, ports, and the study region (see Fig. 2a). Available at the scale of individual MPAs and fisheries, this information was used by the RSG to refine their MPA proposals in subsequent rounds, and by the BRTF to provide guidance on those refinements. Second, maximum potential first order economic impacts, effectively “worst case” scenarios, were provided to stakeholders and the BRTF, at the scale of individual proposed MPAs, port group (Fig. 2b), and study region scales. This information assumed that fishing grounds are lost forever, and that there is no dynamic adjustment to closures, and can be interpreted as the immediate cost of implementing MPAs. Third, starting with the third South Coast region (SCSR), the static analysis provided an assessment of outliers, fisheries likely to experience disproportionate impacts, measured in terms of significant deviation from other values in the sample (Fig. 2c). And fourth, graphs showing a summary of maximum potential impacts across rounds of proposals were provided, by way of illustrating the convergence or divergence between subsequent rounds and variations of MPA proposals (Fig. 2d).

2.3. Dynamic analysis

Two different bioeconomic models were in use for most of the MLPA study regions, although the identity and details of the two models changed somewhat from region to region. A summary of the evolution of the bioeconomic models from the first study region (CCSR) to the final one (NCSR) is shown in Fig. 1. The general form of each model is described by White et al. (2010b) and Rassweiler et al. (2012), respectively; detailed methods and mathematical details of the implementation are given in the Appendix. In brief, both models were single-species, spatially explicit, deterministic age structured models that were parameterized and run to equilibrium for a suite of 6–8 fish and invertebrate species under three different levels of exploitation (outside MPAs) for each proposed MPA network design under evaluation. Larval dispersal among model cells was described by connectivity matrices derived from homogeneous dispersal models in the first two study regions, and Lagrangian drifter simulations in ocean circulation models of the coastal ocean in the last two study regions (Drake et al., 2011; Mitarai et al., 2009; Watson et al., 2010). The results of the two models were quite similar, and hereafter we focus on results generated for the SCSR by the model developed by White and colleagues.

A key uncertainty in the dynamic evaluations was the future fishery management conditions outside of MPA boundaries. The intensity of fishing outside MPA boundaries has a strong effect on population persistence and biomass inside MPAs and yield outside (e.g., White et al., 2010b) but fishery management considerations...
were explicitly separate from all decision-making within the MLPA process (Fox et al., in this issue-b). As such, all model evaluation results were necessarily contingent upon whatever fishery management decisions are made in the future. To represent this uncertainty, model results were calculated for three different fishery management scenarios: MSY-type management (the harvest rate that would achieve maximum sustainable yield [MSY] in the absence of MPAs), conservative management (a harvest rate much lower than that associated with MSY), and unsuccessful management (the population would collapse in the absence of MPAs).

In order to predict spatial changes in biomass and catch over a fixed time period (e.g., 5–10 years) in response to MPA implementation, it is necessary to know what the current, baseline distribution of biomass and fishing effort is. Such data were not available, so instead the bioeconomic models were used to project the spatial distribution of catch and abundance at the long-term dynamic equilibrium (because the equilibrium does not depend on starting conditions). The equilibrium biomass for each MPA proposal was compared to the equilibrium biomass in a model run on starting conditions. The equilibrium biomass and catch was compared to catch in a model run with no MPAs and fishing rate corresponding to MSY. Thus both biomass and yield were expressed as nondimensional proportions of a theoretical maximum.

The dynamic analysis generated two types of results. The first type was spatially explicit model outputs, mainly used to inform the refinement of MPA network proposals between rounds of evaluations in the planning phase. These outputs included maps showing the distribution of suitable habitat (Fig. 3a), the spatial distribution of biomass within the study region for each model species under each fishery management scenario (Fig. 3b), and fishery yield (not shown). These spatially explicit data were also used to calculate several statistics that were useful in evaluating the relative effectiveness of individual proposed MPAs at increasing biomass (see Appendix A), and thus which proposed MPA configurations should be included in revised network proposals. An example of this type of statistic is predicted larval production, which indicated whether a particular proposed MPA successfully increased the export of larvae to elsewhere in the study region (Fig. 3d). An in-depth analysis of these model outputs and their dependence on MPA network configuration and species life history is beyond the scope of this paper, but the results followed the general patterns described elsewhere (Botsford et al., 2001; Moffitt et al., 2009; White et al., 2010a, 2010b).

The second type of result was ranking each MPA proposal in each evaluation round to compare the overall performance of each metric. The proposals were ranked in terms of the mean biomass in the study region (relative to the unfished state, averaged across all model species) and the mean fishery yield in the study region (relative to maximum sustainable yield in the absence of MPAs, averaged across all model species). Both statistics were calculated for each of the three fishery management scenarios (Fig. 4), and were used as overall measures of “conservation value” and “economic value”, respectively.

When overall mean biomass and overall mean yield for a range of MPA proposals in the same study region were plotted on the same axes, they produced a simple pattern that was exhibited by proposals in all study regions as well as models of idealized coastlines with evenly spaced MPAs (Fig. 4; White et al., 2010b). In the unsuccessful management scenario, both biomass and yield were predicted to be higher in MPA proposals with more and/or larger MPAs. Conversely, when the fishery was well managed (MSY-type and conservative management scenarios), biomass and fishery yield were predicted to have a negative correlation across proposals (see Hastings and Botsford, 1999; Mangel, 1998; White et al., 2010b).

2.4. Development and application

It is important to note that both the static and dynamic analysis were developed “in real time”, and responding to the
needs of decision-makers and stakeholders in the process. As such both approaches provide an example of the “bricolage” nature of the use of science in the MLPA process, by which we mean, following Levi-Strauss (1967), a combination of resourcefulness, improvisation and adaptation of emerging methods to the context in which the models were being used (Table 2). Both methods evolved from a relatively basic framework in the first study region (CCSR), eventually incorporating more sophisticated techniques (e.g., inclusion of ocean circulation models in the bioeconomic model) and study-region-specific details (e.g., accounting for recreational fisheries as well as commercial harvest in the static analysis). As the methods evolved, they also gained increasing acceptance by stakeholders, scientists, and decision-makers, and were more integrated into the planning and decision-making process.

3. Comparison of the two analyses

The static and dynamic analyses both attempted to represent the effects of closing fishing grounds on overall fishery yield and/or profit in the study region, although the two analyses focused on different aspects of the issue (summarized in Table 2). The former focused on short term economic effects, while the latter projected population dynamics, for a variety of sheries as well as commercial harvest in the static analysis). As the methods evolved, they also gained increasing acceptance by stakeholders, scientists, and decision-makers, and were more integrated into the planning and decision-making process.

The results in Fig. 5 are shown for each of the three levels of fishing used in the bioeconomic model, which has a strong effect on the relationship between the metrics. The two analyses produce very similar and strongly correlated results under the “conservative” (Fig. 5c) and “MSY-type” (Fig. 5b) fishery scenarios, but were negatively correlated for the “unsuccessful” management scenario (Fig. 5a). This result illustrates a key difference in the mechanisms represented and the outcomes produced by the two methods.

The static analysis requires the assumption that the only effect of MPAs on fishery harvest is to close particular fishing grounds, eliminating the potential to harvest the fish found there. Thus MPAs can have at best a neutral effect on fishery yields in that analysis, and proposals with larger MPAs generally have greater negative effects. This relationship appeared to match many fishermen’s and policymakers’ expectations for the effect of MPAs on their yields (Suman et al., 1999). By contrast, the dynamic bioeconomic model was able to account for the expectation that fish biomass will accumulate inside MPAs and spill over into adjacent fished areas, a pattern that has been shown to occur in many existing MPAs (Halpern et al., 2010). When fishing outside the MPAs is sufficiently intense (e.g., the overfished unsuccessful management scenario), such spillover can be adequate to offset and actually exceed the yield lost by not overfishing in the MPAs themselves. Thus proposals with greater MPA area have higher fishery yields, a result that could not be demonstrated in the static analysis, but which correspond to the expectations of many conservation advocates (Suman et al., 1999). Alternatively, when the fishery is managed...
well enough that MPAs are not necessary for population persistence (the conservative and MSY-type management scenarios), protecting more area in MPAs simply reduces fishing opportunities, as in the static socioeconomic analysis (Fig. 2; White et al., 2010b).

The differing relationships between MPA area and fishery yield in the two analyses under the unsuccessful management scenario also related to the time scales over which those results should be interpreted. Essentially, the static analysis is focused on the immediate effects, whereas the dynamic modeling accounts for the time lags associated with the biological response to the implementation of MPAs. Simply put, it takes fish some time to grow larger and produce more offspring, which can eventually lead to increased population densities both inside and outside the MPA.

It is notable that the results of the two analyses exhibited such a consistent relationship under the two lower-fishing scenarios, especially given the very different levels of sophistication in the two approaches. In particular, the valuations of fishery grounds in the static analysis directly include a range of economic factors affecting the profitability of fishing in a particular location: fuel costs, travel time, fishing strategy, etc., but did not include the ecological factors that would affect the number of fish in a location. The dynamic analysis explicitly represented those ecological factors, but did not include the same economic factors, instead simply assuming that fishing effort would be allocated to patches to maximize average catch per unit effort. Nonetheless, the results in Fig. 5 suggest that at the scale of an entire study region, predictions of overall economic impact are relatively robust to the details of the method, and depend primarily on total MPA area.

4. Use of analyses by scientists, stakeholders, and decision-makers

4.1. Evolving perceptions of both analyses

When they were first introduced into the MLPA process in the CCSR, both the static and dynamic analyses came under intense scrutiny from stakeholders and scientists alike. Eventually, both methods became accepted parts of the process and effort shifted from explaining and defending methodology to improving the accessibility of results to stakeholders. Initial skepticism about the use of these analyses arose from two sources: the lack of explicit mention of economic impact or fisheries in the Act itself (Fox et al., in this issue-b) and the mathematical complexity of the analyses.

The first obstacle, the absence of direct economic goals in the Act, was overcome by the desire of stakeholders and policymakers (i.e., the BRTF) to understand the potential economic and fishery consequences of proposed MPAs. This led to inclusion of the static analysis in the first study region (CCSR) (see Fig. 1). The dynamic model results were not formally included in SAT evaluations until the second study region (NCCSR) after Walters et al. (2007) published an independent model of fish population dynamics in the CCSR that suggested MPAs would be completely ineffective in that region (see Moffitt et al. (2009) for a critique of their analysis). The conclusions of Walters et al. (2007) reinforced both stakeholders and the MLPA initiative the need to include an evaluation that explicitly considered fishery and population dynamics. In particular, the Walters et al. model and subsequent presentations of the bioeconomic model results with conservative fishery management were popular with fishermen because they showed the potential negative effects of MPAs on fishery yield. By the later study regions, fishermen and other stakeholders were strong advocates for the use of both forms of economic analysis because those were the only two science evaluations that addressed the potential economic impacts of the proposed MPAs.

The second obstacle to acceptance, that of mathematical complexity, was also gradually overcome. Both evaluation techniques were treated with some skepticism by members of the SAT and the BRTF in early study regions, but with repeated vetting the SAT eventually accepted the methodologies. For example, a persistent challenge was the perception that because the bioeconomic models relied upon a set of assumptions, they were necessarily untrustworthy. Any mathematical model is based on the premise that complex interactions in natural systems can be described usefully by a set of relatively less complex mathematical expressions (e.g., “assume that fish size can be described by an asymptotic equation fit to size and age data”). As such, explanations of model details to the SAT frequently revolved around the consequences of key model assumptions. Unfortunately, the lay connotation of “assumption” implies a potentially false, reckless premise that is perceived to be inferior to “data” or the lay person’s own anecdotal observations. This misunderstanding led models to be dismissed by many of the public as “too complicated” and/or “based on too many assumptions”. The irony of this reaction is that model opponents typically had their own firmly held expectation of MPA effects, and that expectation was necessarily based in some sort of simple verbal model that itself involved implicit assumptions. An example is the statement that MPAs do not protect fish because fish move around too much. Such verbal models require at least as many implicit assumptions as mathematical models, but fail to state those assumptions explicitly. Mathematical models allow the assumptions to be independently evaluated and help ensure that they are applied consistently and precisely. This type of misunderstanding is not unique to the MLPA process (Caswell, 1988), but our experiences revealed the care that is needed to show how
Table 2
Comparison of key approach and assumptions of the static comparative socioeconomic and dynamic bioeconomic model evaluations.

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<thead>
<tr>
<th>Feature</th>
<th>Socioeconomic analysis</th>
<th>Bioeconomic analysis</th>
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<tr>
<td>System response to MPAs</td>
<td>Static (immediate loss of fishing opportunity)</td>
<td>Dynamic (fishing patterns shift, biological populations grow)</td>
</tr>
<tr>
<td>Computational approach</td>
<td>Field surveys</td>
<td>Numerical simulation</td>
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<tr>
<td>Data sources</td>
<td>Closed-form equations</td>
<td>Published literature</td>
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<tr>
<td>Species included</td>
<td>All commercially significant species</td>
<td>Selected fish species with necessary parameter values</td>
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<tr>
<td>User groups included</td>
<td>All user groups</td>
<td>Representative fishing fleet targeting modeled species</td>
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<tr>
<td>Relevant time scale</td>
<td>Short-term</td>
<td>Long-term dynamic equilibrium</td>
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<tr>
<td>Valuation scheme</td>
<td>Gross and net ex-vessel revenues</td>
<td>Biomass yield as a proxy for gross revenue</td>
</tr>
<tr>
<td>Accounts for biological connectivity</td>
<td>No</td>
<td>Larval dispersal (based on ocean currents) and adult movement</td>
</tr>
<tr>
<td>Accounts for human behavior</td>
<td>Based on locations and values reported by users</td>
<td>Based on ideal-free distribution model</td>
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Fig. 5. Comparison of results from comparative static and dynamic bioeconomic analyses for nine MPA proposals from the first round of the South Coast Study Region. Socioeconomic results are expressed as proportional reduction in gross profits, relative to current dollar values (at the time of analysis); bioeconomic results are expressed as proportional reduction in total fishery yield (by mass), relative to maximum sustainable yield (MSY) levels. Results and Pearson’s correlation coefficient (r) are shown for the three fishery management scenarios used in the bioeconomic model (panels a–c; note that socioeconomic results do not vary between panels). Results are shown for four species included in both analyses: kelp bass (Paralabrax clathratus; open symbols), California halibut (Paralichthys californicus; filled symbols), California sheephead (Semicossyphus pulcher; split symbols with dark top), and red sea urchin (Strongylocentrotus franciscanus; split symbols with dark right side). MPA proposals are indicated by marker shape: External A (circle), External B (upward triangle), External C (square), Lapis A (diamond), Lapis B (right triangle), Opal A (left triangle), Opal B (down triangle), Topaz A (six-pointed star) Topaz B, (five-pointed star).

Quantitative models are preferable to the alternative simple verbal model.

Both methods also confronted a running debate regarding the accessibility of their methodologies and data to stakeholders and other scientists. For example, in the static analysis, data accessibility was limited because of the sensitive nature of the data used in the analysis: individual fishermen did not want to reveal their preferred fishing grounds or have proprietary business information become part of the public record. In general these concerns were met by following standard confidentiality protocols, with data presented in aggregated form to protect individual information. In the case of both approaches, all model results, equations and parameter values were made publically accessible to encourage review by stakeholders and scientists. However, the actual bioeconomic model codes were never released to the public, despite some requests to that effect. This decision pitted the desire to allow stakeholders to actually use the models in the design process against the lack of time and funds to either train lay users or create a lay-friendly user interface.

4.2. Concerns and critiques by stakeholders, scientists, and policymakers

Over the course of the MLPA process, several criticisms of both analyses arose repeatedly but could not be resolved within the constraints of the process itself. For example, the list of species represented in the bioeconomic model was controversial in each study region. Fishermen typically advocated including the full range of commercially and recreationally important species in the region in order to capture all potential reductions in catch. This was usually not possible because key demographic parameters were not available for some of those species, or the species had life histories incompatible with the modeling framework (e.g., large scale spawning migrations, or use of estuarine nurseries). This contrasts with the static socioeconomic analysis, which was able to include all commercially significant fish species. Other stakeholders and scientists consistently advocated for the inclusion of well-studied or charismatic species that were not fished. This request typically arose from the expectation that MPAs are beneficial for all marine organisms, not just fishery species. However, the only effect of MPAs in the dynamic model was to remove fishing, and there was no mechanism in the model to produce other potential MPA effects such as improved habitat quality or food web complexity. Thus models of non-target species would have shown no difference between MPA proposals, and were not pursued.

A related criticism had to do with the alternative fishery management scenarios used in the bioeconomic model. Many fishermen argued that because some nearshore fisheries were currently undergoing rebuilding plans with heavy restrictions on harvest (e.g., rockfish; Gleason et al., 2006), only the “conservative management” results — which showed consistently negative effects...
of MPAs on yield (Figs. 4 and 5) – should be used. Indeed, some fishermen (as well as some fishery biologists) took offense at the implication that they would allow a stock to be overfished. This complaint represented a misunderstanding of the management scenarios, which were intended to represent conditions at equilibrium, potentially many years in the future. It was not appropriate to “plug in” current fishery management policy for a given species, except insofar as the recent management history indicates the likelihood that a species will be successfully managed in the future (see White et al., 2010b). With respect to the static analysis, stakeholders frequently voiced criticisms of the representativeness and comprehensiveness of the data being used. In particular, the purposive sampling approach was criticized for not being rigorous enough to produce statistically significant results commonly associated with random sampling. This charge stems from a misunderstanding of statistical methods, and also missed the point that the population affected by the MLPA implementation was both well known and a subset of the larger California citizenry. A related complaint was that the sampling was not comprehensive enough. Ironically, this complaint often arose because some fishermen groups declined to participate in the survey process, which then led to complaints by them and other fishermen that the socioeconomic data were incomplete. This was addressed with reporting the sampling success in particular fisheries and ports.

Finally, both approaches faced criticism of being too dispasionate, apparently in reaction to the impartiality with which models and results were presented that were reporting impacts on people’s livelihoods. In an apparent response to these considerations, the BRTF eventually requested an analysis of outliers, i.e., fisheries and ports that would be disproportionately affected by an MPA proposal. While the MLPA does not contemplate compensation mechanisms per se, policy-makers wanted to ensure that individual fishermen or ports were not taking the brunt of the impacts. Anecdotal evidence further suggests that the outlier information has formed the basis for arrangements to involve affected fishermen in monitoring activities.

4.3. Relationship to other evaluation metrics

While the comparative static analysis stood alone as the only science evaluation of MPA proposals used in the MLPA process that expressed results in dollar values, the estimates of conservation value produced by the dynamic bioeconomic modeling were similar in intent to several of the science guidelines that were designed to ensure compliance with the stated goals of the Act (see Kirlin et al., in this issue, for a full description of these goals). The habitat representation, size, and spacing guidelines were all intended to ensure that MPAs were the correct size and configuration to support viable populations (MLPA goal 2) and operate as a biological network (MLPA goal 6; see Saarman et al., in this issue for a description of the guidelines). The size and spacing guidelines were originally based loosely on estimates of larval and adult dispersal distances (e.g., Kinlan and Gaines, 2003), and resulted, as a matter of mathematics, in a requirement that a fraction of coastal habitat be covered by MPAs (although no such requirement was included in the MLPA itself). This relatively rigid type of guideline contrasted with the dynamic analysis, which allowed a direct calculation of population persistence that included the variation in habitat availability and larval dispersal distances (among other factors) present in the actual study region. Despite the opportunity for more precise calculations of population persistence afforded by the bioeconomic models, the size and spacing guidelines were never refined. This situation arose for a variety of reasons. From a non-scientific standpoint, the size and spacing guidelines were introduced earlier in the MLPA process and became codified in the Master Plan document that governed the process (Kirlin et al., in this issue; Saarman et al., in this issue). While contemplated by the Act, it became infeasible to revise the Master Plan during the MLPA process, effectively delaying the integration of emerging scientific results and approaches into the decision making process. The size and spacing guidelines were also viewed as simpler and more reliable by many stakeholders and scientists than assumption-laden dynamic models (despite the many assumptions underpinning the size and spacing guidelines themselves; see Section 4.1 and Saarman et al., in this issue), and perhaps more representative of species not directly modeled in the bioeconomic analysis. Additionally, the size and spacing guidelines could provide specific targets for MPA design whereas the bioeconomic model simply compared proposals with alternative designs. Nonetheless, in practice relatively few MPA proposals met all of the size and spacing guidelines, and those that did typically met only the minimum requirements (minimum size or maximum spacing allowed; Saarman et al., in this issue). Because such guidelines are based on strict thresholds, there was no advantage to propose MPAs that were bigger or more closely spaced (although the BRTF did urge the RSG to propose MPA networks that exceeded the minimum recommendations). This is unfortunate as Moffitt et al. (2011) later showed that spatially explicit models allow greater flexibility in achieving goals than fixed size and spacing recommendations. For example, increasing MPA size beyond the size guideline could produce a linear increase in the number of species protected, whereas reductions in spacing below the maximum guideline yielded minimal additional benefit.

Ideally the size and spacing guidelines and bioeconomic model could have been used more successfully in tandem. Given the legal framework and the design of the MLPA process, in which all MPA proposals originated with stakeholders (Kirlin et al., in this issue), it would have been cumbersome to use the models to optimize management decisions (see Section 5.1). In a different institutional setting, an alternative decision-making process might be to use size and spacing guidelines as recommendations to guide the initial design process, and then refine the initial proposals using the more precise evaluations afforded by bioeconomic models.

5. Discussion and conclusions

The comparative static analysis and dynamic bioeconomic model were distinct but complementary tools for economic analysis in the MLPA process. Formal economic assessments were not mandated by the MLPA, but in the later study regions both of these analyses were being utilized by stakeholders, scientists, and policy-makers. The static analysis was essentially a first-order, worst-case representation of the potential economic loss imposed by closing off fishing grounds in MPAs. It provided a counterbalance to the largely ecologically-based science guidelines, which favored proposals with large MPAs without regard for potential economic impacts (Saarman et al., in this issue; Gleason et al., in this issue). Similarly, the bioeconomic analysis provided a quantitative representation of the likely tradeoff (or lack thereof, depending on the fishery management scenario) between achieving conservation versus economic goals. Although not linked directly to catch data, it did capture the potential for spillover from successful MPAs into fished areas and the resulting increase in fishery yields.

5.1. Comparison to MPA network design efforts elsewhere

The development of both static and dynamic economic analyses described here was both cutting-edge and somewhat unique in the world of MPA planning. This is primarily due to the decision-making structure of the MLPA process, in which MPA boundaries

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were proposed by stakeholders, evaluated by scientists and then considered by policymakers (Kirlin et al., in this issue). Globally, a large proportion of MPAs are placed out of political convenience rather than a scientific design process (e.g., Francour et al., 2001; Idechong and Graham, 1998; Monaco et al., 2007; Russ and Alcala, 1999; Walls, 1998). Others are designed to minimize socioeconomic impacts, often using a survey of local users to estimate costs, as in the comparative static approach described here (Aswani and Lauer, 2006; Friedlander et al., 2003; Manson and Die, 2001). However, the analyses here were unprecedented in scope, spatially explicit nature, and an effective compilation of data from very different fisheries over a large spatial area to estimate total costs.

Of MPA design approaches that address both economic and biological goals, the most common is an optimization strategy in which economic impacts are minimized while achieving a minimum biological threshold. For example, the simulated annealing software MARXAN (Possingham et al., 2000) has been applied to MPA design problems across the globe (e.g., Harris, 2007; Green et al., 2009; Klein et al., 2008a). A limitation of such static optimization approaches is their implicit assumption that protecting some minimum fraction of coastal habitat will ensure the persistence of the various biological populations of interest (Possingham et al., 2000). Unfortunately, the size and configuration of MPAs required to ensure persistence depends on many factors and can only be calculated using dynamic models (Botsford et al., 2001; Kaplan et al., 2009; Pressey et al., 2007; White et al., 2010a). To date several types of bioeconomic models have been used to demonstrate ideal MPA configurations for particular locations (e.g., Beattie et al., 2002; Costello et al., 2010; Little et al., 2007; Salomon et al., 2002; Stockhausen et al., 2000). However, to our knowledge the work reported here is the first example of a dynamic model being used as a formal, real-time component of an MPA network design process.

Optimization-based MPA design was not possible within the stakeholder-driven MLPA process, because scientists were not charged with proposing MPA configurations and the MLPA lacked formal economic goals (Fox et al., in this issue-a, -b; Kirlin et al., in this issue). However, later analyses showed that MARXAN-optimized designs could have produced lower costs to stakeholders (in terms of closed fishing grounds) than the configurations proposed by the stakeholders themselves (Klein et al., 2008b, 2010b). Indeed, the power of the economic analysis tools described here was limited by the narrow scope of alternative configurations that could be proposed by stakeholders who were constrained in their designs by limited habitat and size and spacing guidelines. Neither tool could reveal biologically or economically preferable configurations when applied to a small set of very similar proposals. For example, in the SCSR, the initial suite of proposals was very broad in terms of total habitat protected, but relatively uniform in terms of MPA locations, with less conservation oriented proposals typically just subsets of more extensive proposed networks. Even the variation in habitat protected shrank considerably in the second round to a smaller set of options that all produced similar results in the bioeconomic model (compare round 1 and round 2 proposals in Fig. 4). The possibility of low variability among MPA proposals limiting the range of bioeconomic model outputs (and thus the utility of the model in discriminating among proposals) should be considered when applying quantitative tools to future stakeholder-driven processes.

5.2. Lessons learned

While the implementation process has concluded, it is too early to tell whether the resulting MPAs will meet the goals of the Act and what contribution the economic analyses described here made to that success. With these considerations in mind, we focus on lessons related to the MPA design process, not MPA performance.

5.2.1. Combining MPA design and conventional fishery management

The most striking result of the dynamic bioeconomic modeling was that fishery management outside MPAs had a greater effect on predicted change in equilibrium biomass in a region than did the size or configuration of the MPAs themselves (see also Hastings and Botsford, 1999; Mangel, 1998; White et al., 2010b). Indeed, the fishing mortality rate outside proposed MPAs determined whether increasing the area in proposed MPAs would produce tradeoff between biomass and fishery yield (relatively low fishing) or will increase both quantities (very high fishing). Consequently, it was difficult to predict the likely effects of a particular MPA proposal without being able to specify what fishery management would be like in the future: a particular proposal could produce the highest or the lowest fishery yields relative to other proposals, depending on the management scenario. A possible solution (though not one available within the MLPA legal framework) would be to combine MPA design and conventional fishery management into a single process in order to jointly set target outcomes. Of course it should be noted that MPAs also have indirect effects on non-fishery species (e.g., Shears et al., 2008); for which this point may be irrelevant; additionally, the existence of no-take areas can either complicate or improve the monitoring involved in conventional management (Babcock and McCall, 2011; Field et al., 2006).

5.2.2. Organizational inertia: spatial models versus size and spacing guidelines

One of the questions future implementation efforts will have to address is how and whether to utilize static design guidelines (such as the SAT’s size and spacing guidelines, Section 4.3; Halpern et al., 2006; Palumbi, 2004; Shanks et al., 2003) versus dynamic modeling approaches. With non-experts creating the initial MPA proposals (as in the MLPA process), it may be more expedient to use static guidelines than to use models. However, such guidelines can carry latent policy implications, such as a target fraction of habitat to be protected, which is implicitly determined by size and spacing guidelines. This complicates the role of scientists in MPA planning processes, whose role some have posited is to provide decision-makers with predictions of the consequences of having certain amounts of area in MPAs (Moffitt et al., 2011; White et al., 2010b), not to make the decision regarding the amount.

The tension between models and guidelines in the MLPA process highlights a potential pitfall of this type of multi-stage process, especially one dependent on the “best readily available science” (Kirlin et al., in this issue; Saarman et al., in this issue). Simple rules-of-thumb such as the size and spacing guidelines were the best available approach before the models were developed and became available to the process. Given the implementation schedule, however, it proved infeasible to revise the MLPA Master Plan. Consequently, even as the science improved, the initial guidelines remained part of the process. This inertia was reinforced throughout the process by the continued skepticism of the dynamic models by some stakeholders, scientists, policy-makers, and decision-makers.

5.2.3. Combining static and dynamic economic analyses

One missed opportunity in the MLPA process was the failure to successfully use the socioeconomic data used in the static analysis to parameterize a realistic sub-model of fisherman behavior for use in the dynamic, bioeconomic model. The current state of the art in such efforts are so-called “discrete choice” models in which fishermen are assumed to make a decision about which patch to fish in each time step based on a series of factors including knowledge of
past catch-per-unit-effort in that patch, distance from port, weather, etc (e.g., Smith et al., 2010). Wilen et al. (2002) developed such a model for the red sea urchin in northern California using catch data and an existing metapopulation model. The MLPA process afforded an opportunity to expand that sort of model to a much greater level of spatial detail and oceanographic realism for multiple species, but we lacked the time and resources to undertake that effort. However, given the potentially counterintuitive results suggested by Wilen et al.’s (2002) analysis, including data-driven models of fishermen behavior is a valuable undertaking.

5.2.4. Factors missing from the analyses

Although the bioeconomic models were run for multiple model species, they were fundamentally single-species models. That is, they did not include interactions among species. There can be great value in directly representing how interspecific interactions may shape the effects of management decisions on different members of the ecological community (Crowder and Norse, 2008; Shears et al., 2008), and ecosystem modeling approaches have been used to that end (e.g., Salomon et al., 2002). The advantages of ecosystem models come at the cost of greater data needs for parameterization and sometimes problematic structural assumptions (Plagángyi and Butterworth, 2004), and so we did not pursue that alternative.

By necessity, the static analysis only considered direct impacts on commercial and charter boat fisheries. It remained silent on indirect and induced effects, the so called “multiplier” effect resulting from the initial changes income on other businesses in the coastal economies of California. This was a source of some

Table 3

Data required to implement (a) our static economic analysis and (b) our dynamic bioeconomic analyses in a new location. We provide both the ideal type of data (as used in this paper, see main text and Appendix for details) and a low-information substitute for data-poor locations (including some examples of their use).

<table>
<thead>
<tr>
<th>Data type</th>
<th>Preferred</th>
<th>Potential substitutes</th>
</tr>
</thead>
<tbody>
<tr>
<td>a. Static economic analysis</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Spatial fishing patterns (extent and intensity of use)</td>
<td>• Direct surveying of a representative sample population of fishermen</td>
<td>• Logbook data based on landing blocks</td>
</tr>
<tr>
<td>• Spatial data for each port-fishery combination</td>
<td>• Spatial data at the regional fishery level</td>
<td></td>
</tr>
<tr>
<td>Landings and ex-vessel revenue data</td>
<td>• Landing receipts at the individual fisherman level</td>
<td>• Landing receipts at the port or regional level</td>
</tr>
<tr>
<td>Operating costs and regulations</td>
<td>• Tax documents or other official financial records</td>
<td>• Self reporting from fishermen</td>
</tr>
<tr>
<td>• MPA boundary coordinates and specific fishery regulations</td>
<td>• Assume MPAs are no-take for all species</td>
<td></td>
</tr>
<tr>
<td>b. Dynamic bioeconomic analysis</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Life history information</td>
<td>Sufficient data to estimate Lifetime Egg Production:</td>
<td>• Equivalent data for taxonomically or ecologically similar species (e.g., from FishBasea)</td>
</tr>
<tr>
<td>• age—length relationship (e.g., von Bertalanffy curve)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• length—fecundity relationship</td>
<td></td>
<td></td>
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<tr>
<td>• age at maturity (or maturity ogive)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• age or length of entry to fishery</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• natural mortality rate</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Habitat</td>
<td>• Quality and density of suitable habitat for each species in each spatial model cell</td>
<td>• Presence/absence of suitable habitat within a model cell (e.g., White et al., 2010a,b)</td>
</tr>
<tr>
<td>Larval dispersal patterns</td>
<td>• Dispersal matrix created by estimating probability of larvae dispersing from each model cell to any other model cell, derived from Lagrangian simulations (including appropriate larval behaviors) within flow fields generated by a validated ocean circulation model</td>
<td>• Assume subtidal habitat is similar to adjacent intertidal habitatb</td>
</tr>
<tr>
<td>Adult movement</td>
<td>• Fine-scale estimates of movement from acoustic tagging studies</td>
<td>• Estimates of mean dispersal distance from genetic or natural larval tag studies (reviewed by Botsford et al., 2009)</td>
</tr>
<tr>
<td>MPA locations &amp; regulations</td>
<td>• Resolution of population model matches size of smallest MPAs</td>
<td>• Spatially homogeneous dispersal kernel estimated using PLD, flow information, and maps of coastal topography and bathymetry (e.g., Fischer et al., 2011)</td>
</tr>
<tr>
<td>• Species-specific regulations within the MPA</td>
<td>• Assume MPAs are no-take for all species</td>
<td></td>
</tr>
<tr>
<td>• Estimates of poaching ratesc</td>
<td>• Assume fractional protection within a model cell if MPAs occupy only a portion of the cell</td>
<td></td>
</tr>
<tr>
<td>Initial conditions</td>
<td>• Density and age structure in each model cellc</td>
<td>• Avoid assumptions about initial conditions or current fishing; run simulations to long-term equilibrium (this paper)</td>
</tr>
<tr>
<td>• Current, pre-MPA fishing mortality ratec</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Future fishery management</td>
<td>• Range of potential fishing mortality rates (and spatial distributions of fishing effort) outside MPA boundariesc</td>
<td>• Simulate dynamics under a wide range of potential fishery scenarios (from overexploitation to minimal exploitation), use a decision analysis framework (White et al., 2010a,b; this paper)</td>
</tr>
</tbody>
</table>

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b. This approach was used in some cases in the NCSR when nearshore subtidal habitat data were unavailable.
c. Not included in the analyses described here.

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frustration for stakeholders and decision-makers, both of whom called for more and more comprehensive economic data. The analyses also refrained from considering the (potentially significant) costs associated with enforcement of MPA regulations (e.g., see Halwood, 2005); this was primarily because the agency responsible for enforcement specifically requested that enforcement cost considerations not affect deliberations within the MLPA process.

Both analyses were strictly in the realm of economic theory and practice, with the static analysis including a minimal socioeconomic characterization of the various fisheries investigated in terms of operating costs and basic demographics. There is thus very little “social” in the socioeconomic analysis, and a more comprehensive assessment of MPA impacts on coastal users and communities would include historical, institutional, cultural, and sociological data and analysis. Including this greater context in static and bioeconomic analyses could inform how their predicted impacts may affect and alter the behavior of particular stakeholder groups. These insights, in turn, can better inform understanding of where resistance to MPAs may come from and why.

Despite these limitations, the SAT consistently agreed that these analyses represented the “best readily available science” given the available input, resources, and data. Regardless, the pace of formal peer review and journal publication is far slower than that of the MLPA process, so these methods have been formally reviewed and documented only after they were used in management decisions. This was a frequent subject of complaints by skeptical stakeholders and scientists who wished to see greater peer review, but was inescapable given the deadlines and rapidly involving methodologies involved (Table 2).

5.3. Moving forward

5.3.1. MPA assessment in California

Now that MPAs have been designed and implemented in each of the MLPA study regions, the process is moving into the monitoring phase of implementation (Gleason et al., in this issue). Monitoring is an essential part of adaptive management, an operating principle that underpins most implementations of MPAs (Grafton and Kompass, 2005), including the MLPA (Carr et al., 2011; Gleason et al., in this issue). At this early stage we can say little about the success of these monitoring efforts but we can underscore why monitoring is important (White et al., 2011). Examining the outcomes of our modeling efforts, we see a substantial dependence on uncertain parameters and states, for example larval dispersal distances and the intensity of fishing (White et al., 2010b; White and Rogers-Bennett, 2010). Similarly, most meta-analyses of MPA monitoring data indicate that there is considerable variability in MPA effects (e.g., Halpern, 2003; Lester et al., 2009). While on average most MPAs have produced increases in biomass, some have led to no change or even a decrease in biomass (Lester et al., 2009; White et al., 2011). Unfortunately, simply observing that a given MPA has not produced an increase in biomass does not reveal what went wrong: was the MPA too small, was enforcement insufficient, was too much fishing allowed outside the MPA, or has insufficient time elapsed to observe an effect? Careful use of models in concert with monitoring data could assist in resolving that question (Pelletier et al., 2008; White et al., 2011).

Both the static and dynamic approaches described in this paper could play a role in upcoming adaptive management efforts. The static socioeconomic analysis provides considerable data on base-line conditions at the time of MPA implementation as well as a first-order prediction for what the initial effects of MPAs will be on the fishery. This dataset could also be reanalyzed to predict the real-location of fishing effort after implementation: in theory the fishing fleet should shift effort to the next best set of fishing grounds closest to the high value areas now in MPAs. This information could prove valuable in calibrating the expected level of fishing outside MPAs within the bioeconomic model framework. At present, the bioeconomic model is focused solely on the long time scales associated with deterministic equilibria. As baseline data become available from monitoring efforts, the model can be shifted to operate over shorter time scales and interact with the monitoring data (White et al., 2011; White and Rogers-Bennett, 2010).

5.3.2. Beyond California

The idea that fishing outside of MPAs can affect processes within MPAs (Section 2.3) suggests that MPAs cannot be managed independently but rather should be part of a coast-wide management effort. Indeed, resource managers and conservation scientists have already begun moving away from a strictly MPA-based approach to conserving marine resources, instead favoring the concepts of Integrated Coastal Management (Cicin-Sain and Belfiore, 2005) and Marine Spatial Planning (Crowder and Norse, 2008). Within that framework, the analytical tools developed here would need to be expanded to include not just fishing, but other uses, such as recreational diving or wave energy (e.g., Klein et al., 2010). Some nations have already begun implementing large scale zoning plans without the benefit of the economic analyses described here (e.g., the Great Barrier Reef: Day, 2002, 2008; Olsson et al., 2008), but such tools could only improve the process. To that end, we provide a table outlining the required data and other types of information needed to implement our analyses in other regions, as well as potential substitutes when lack of time or resources preclude the collection of preferable data types (Table 3). We hope that scientists and managers worldwide are able to adapt and improve upon our techniques in order to improve the spatial management of marine resources.

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Ethical statement

We declare that we have no conflicts of interest.

This work has not been published previously, is not under consideration for publication elsewhere, and if accepted will not be published elsewhere without consent of the copyright holder.

This manuscript has been approved by all authors, and all authors materially participated in the research or article preparation. The work described in this manuscript complied with the laws of California and the United States.

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