A PROPOSED FRAMEWORK FOR IDENTIFYING AND EVALUATING ECOSYSTEM SERVICE TRADEOFFS FOR MARINE PROTECTED AREA PLANNING

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ABSRACT

Existing research on the effectiveness of marine protected areas narrowly focuses on developing sets of management indicators tied to outcomes described in management plans or on the achievement of a single objective such as an increase in the size or number of older, more fecund female fish in a protected area. These approaches, however, fail to capture the dynamic complexity of social, cultural, economic and ecological processes, uncertainty, and tradeoffs associated with the establishment and operation of marine protected areas. Moreover, there is usually little recognition of local coastal communities, including fishing communities, and the values they hold in the evaluation of the relative benefits and costs of marine protected areas. We describe a framework for evaluating and assessing economic and ecosystem tradeoffs associated with marine protected areas and other forms of marine spatial planning. This framework integrates biophysical data and community-based social and economic evaluation methods. It identifies ecosystem services, their benefits and associated uncertainties in assessing the economic and social tradeoffs associated with changes in the use of the marine environment.

Keywords: Ecosystem Services, Marine Protected Areas, Stated Preferences, Trade-off Analysis

INTRODUCTION

Existing research on marine protected area (MPA) effectiveness focuses on developing sets of management indicators tied to outcomes listed in a MPA management plan (see Pomeroy et al. 2005) or on the achievement of a single objective such as an increase in the size or number of female fish in the reserve (see Bernstein et al. 2004). These approaches, however, fail to capture the dynamic complexity of social, cultural, economic and ecological processes and the trade-offs associated with the establishment of MPAs.

There is a need for new approaches to evaluate the efficacy of MPAs that are grounded in the emerging ecosystem-based management (EBM) paradigm. The goal of EBM is to conserve, maintain and restore ecosystem functions to promote the economic and ecological sustainability of marine ecosystems and human communities, both coastal and more broadly, that depend on the services they provide (McLeod et al. 2005; Levin and Lubchenco 2008). Key to integrating the biophysical and human dimensions of EBM is the concept of ecosystem services (Fisher et al. 2009; Crowder and Norse 2008). These can be defined as "aspects of ecosystems utilized (actively or passively) to produce human well-being" (Fisher et al. 2009, p. 645). Ecosystem services have become an important focus in natural resource management as a way of integrating economic and ecological considerations into ecosystem-based decision-making systems (Fisher et al. 2008; Fisher et al. 2009; Millennium Ecosystem Assessment 2005; NRC 2005). Ecosystem services of coastal systems include (Beaumont et al. 2007; Millennium Ecosystem Assessment 2005): (1) Provisioning services that are the direct products obtained from the ecosystem such as fish taken for food; (2) Regulating services, such as the role that extensive kelp beds can play in preventing shoreline erosion; (3) Cultural services providing nonmaterial benefits to humans, such as the identity of a community as a fishing community or a center for whale watching and; (4) Supporting services that are necessary for the production of other ecosystem services, but do not directly benefit humans, such as the habitat structure provided by a rocky reef.

To bring these general tenets from idealism to pragmatism, measurable indicators must be identified, evaluated, and monitored over time. We propose an ecological indicator-based valuation framework that (1) Uses marine ecosystem services and benefits as indicators for estimating economic use and non-use values/benefits associated with marine resources; (2) Engages managers, communities and stakeholders in the evaluation process to incorporate local knowledge and community values and to increase the probability of stakeholder buy-in; and (3) Explicitly takes into account uncertainty when evaluating changes in ecosystem services and associated trade-offs resulting from the establishment of MPAs.

CHARACTERIZING ECOSYSTEM SERVICES AND BENEFITS

A number of different ecosystem services classification schemes have been proposed (see, e.g., Boyd and Banzhaf 2007; Farber et al. 2006; Fisher et al. 2009; Wallace 2007). Although there is much common ground about the nature and importance of ecosystem services, there is less agreement on how to classify these services for resource management and environmental decision-making (Costanza 2008; Fisher and Turner 2008; Wallace 2008).

Following Fisher et al. (2009) we recommend an ecosystem services classification approach that first defines the ecological services being considered and second, defines the human benefits these services generate. For example, an ecosystem service is the production of fish stocks while the benefits are the direct impact on human health and well-being such as food and recreational fishing opportunities (Fisher et al. 2008). Services can be further divided into intermediate services such as primary production or stock recruitment and final services such as food production (Fisher et al. 2008; Boyd and Banzhaf 2007).

There is a very limited literature looking at marine ecosystem services (see, e.g., Beaumont et al. 2007; Holmlund and Hammer 1999; Millennium Ecosystem Assessment 2005; Zheng 2009) and ecosystem services associated with marine protected areas in particular (see Roncin et al. 2008). These studies identify a number of issues that have proved problematic in ecosystem service studies. These include: (1) Ecological complexity including non-linearity in ecosystem functions; (2) Treatment of rival vs. non-rival and excludable vs. non-excludable services associated with MPAs; (3) Joint production where a single ecosystem service can produce multiple benefits; (4) Benefit dependence, where the classification of ecosystem service is influenced by the benefits being considered and; (5) Spatial and temporal dynamics where ecosystem services and benefits are heterogeneous in time and space (Boyd and Banzhaf 2007; Chee 2004; Cowling et al. 2008; Fisher et al. 2008; Fisher et al. 2009; Limburg et al. 2002).

ECOLOGICAL INDICATOR-BASED VALUATION APPROACH FOR ESTIMATING CHANGES IN ECOSYSTEM SERVICES AND BENEFITS

Schiller et al. (2001) define an ecological indicator as an "expression of the environment that provides quantitative information on ecological resources…frequently based on discrete pieces of information that reflect the status of large systems (Hunsaker and Carpenter 1990)." For example, an indicator of marine living resources might characterize the health, composition, or diversity of marine organisms present in an ecosystem. Within stated preference (SP) or survey-based valuation, the role of ecological indicators is to communicate changes in resource quality or quantity, such that meaningful expressions of value may be elicited. Such information must not only be placed in a format that is readily understood by respondents, but that also provides an accurate representation of the policy change being valued (Mitchell and Carson 1989; Bateman et al. 2002). As stated by Schiller et al. (2001), "effective communication of ecological indicators involve[s] more than simply transforming scientific phrases into easily comprehensible words. [It requires] language that simultaneously fit[s] within both scientists' and nonscientists'…frames of reference, such that resulting indicators [are] at once technically accurate and understandable." Furthermore, the validity of welfare estimates depends on appropriately integrating ecological indicators and economic information (Johnston et al. 2002, 2010).

Stated preference methods are frequently employed to assess use and nonuse values associated with changes in environmental resources (Aas et al. 2000; Bateman and Willis 1999; Bauer et al. 2004; Bennett and Blamey 2001; Collins et al. 2005; Nunes and Blaeij 2005; Teisl et al. 1996; Wessells 2002). Nonetheless, unlike indicators developed within the ecological literature (e.g., Karr 1991; Engle et al. 1994; Summers et al. 1995; Weisberg et al. 1997), indicators used within SP surveys are often based on *ad hoc* metrics unrelated to formal models of ecosystem change. More specifically, measures of change in environmental resources presented in SP surveys: a) are rarely developed within the context of established ecological models; b) rarely address uncertainty associated with prediction and measurement, c) are often ambiguously linked to quantifiable and measurable long-term policy impacts; and d) are often based on arbitrary or vague measurement units. Lack of quantifiable correspondence between changes in environmental resources provided in survey instruments and measurable changes resulting from policy implementation may render benefit estimates of limited relevance, lead to biased welfare estimates, and contribute to misguided policy.

A second feature given little attention in the SP literature is the different ways in which the public understands changes in environmental resources and in which such attributes affect utility. Focus groups reveal that respondents typically view ecosystem changes relative to historical or pristine conditions, or based on a common language understanding of ecosystem health (Schiller et al. 2001). In contrast, common metrics used in SP surveys (e.g., 1 million juvenile fish; 20,000 sea birds) typically have little meaning to respondents. Indeed, with the exception of a few well-known metrics (e.g., the RFF water quality ladder, Vaughan 1982), there has been little attention provided to the development of meaningful (Elbert and Welsch 2004), consistent ecological indicators in SP research. The concern is that if survey scenarios provide inaccurate or confusing representations of environmental policy impacts, even the most apparently robust willingness to pay (WTP) estimates may provide biased welfare information.

Our proposed framework estimates theoretically-consistent economic values and models that are widely applicable and transferable, focused specifically on methods to estimate total benefits, including use and nonuse values (Johnston et al. 2010). This will help ensure that ecosystem services valuation studies are based on realistic descriptions of baseline conditions and changes from the baseline. The proposed framework explicitly addresses the correspondence between valuation based on indices and those based on more specific descriptions of ecological services, ecological benefits, or bundles of benefits.

A Conceptual Ecological-Economic Model for Indicator-Based Valuation

The basis for our conceptual model draws from established theoretical linkages between human values and ecological indicators, exemplified by figure 1 below (cf. Schiller et al. 2001). In a contingent valuation or SP context, assessment endpoints (USEPA 1998) often represent unobservable policy goals that influence well-being or affect utility. For example, utility from policies to prevent aquatic species mortality may depend on resulting changes in species health, abundance, and diversity—attributes that are often not directly observable. Direct and indirect indicators are measurement endpoints, or ecological measures that are used within formal frameworks to communicate, infer, or predict changes in assessment endpoints (USEPA 1998).

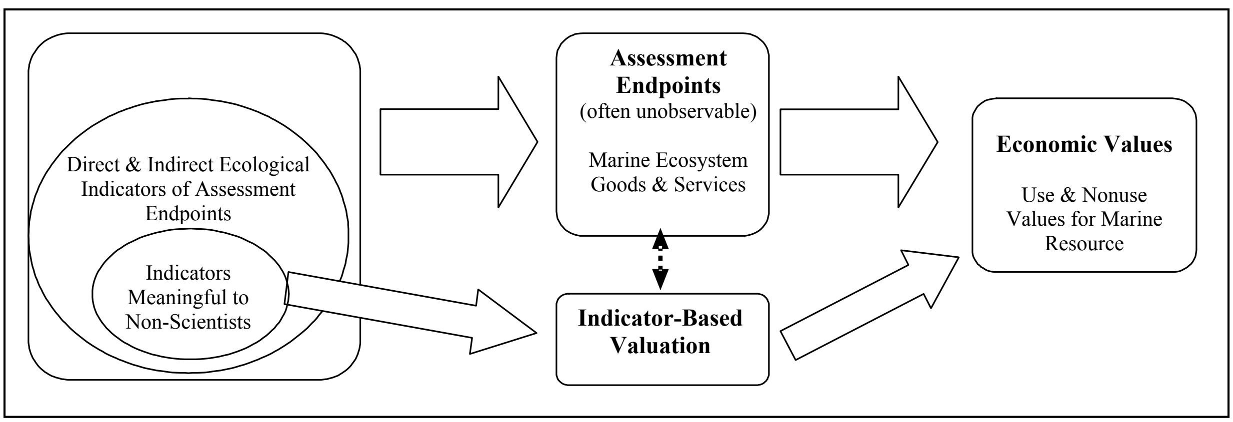


Figure 1. Conceptual Model Linking Values to Ecological Indicators for Marine Resources

SP valuation protocols provide no standardized framework for linking indicators to assessment endpoints to economic values (Johnston et al. 2010). As a result, linkages between assessment endpoints and ecological indicators (i.e., measurement endpoints) in SP surveys are nebulous, leading to ambiguous relationships between ecological change and value. Given the lack of a formal ecological framework guiding SP valuation of use and nonuse benefits, valuation approaches are often at odds with measurable policy outcomes and ecological models. A common consequence is that resulting WTP estimates cannot be easily integrated into benefit cost analyses due to a lack of correspondence to measurable ecological outcomes. For example, use of indistinct measures such as ‘low’, ‘medium’, or ‘high’ classifications for biodiversity (Lupi et al. 2002; Carlsson et al. 2003), or ‘unique’ ecological conditions (McGonagle and Swallow 2005; Opaluch et al. 1993), or ranges in flora and fauna as ‘wide’ versus ‘poor’ (Hanley et al. 2006) are not critically linked to assessment endpoints.

Given these limitations, it is unlikely that the current state-of-the-art SP survey methodology provides sufficient information to enable meaningful WTP elicitation for ecological benefits in most marine policy contexts. For example, focus groups repeatedly reveal that the public shares little understanding of the magnitude, meaning, or relevance of such measures as, for example, 20,000 common sea birds, 1 million juvenile fish, or other measures commonly used in SP instruments (Johnston et al. 2004a,b)—even if such metrics were to correspond to outcomes modeled by natural scientists (which they often do not). While they may indicate that "more is better," focus group participants typically express a general lack of shared understanding of such metrics in terms of general ecosystem status. This fundamental lack of understanding of ecological changes reflected in survey scenarios is a classic example of the type of methodological misspecification discussed by Mitchell and Carson (1989) and Johnston et al. (1995)—and will almost certainly lead to biases in benefit estimation.

The ecological literature provides numerous integrative ecological indices that relate measurable characteristics to both biotic and abiotic ecosystem components, allowing these indicators to be used to quantify relative ecosystem health (Simon and Lyons 1995; Attrill 2002; Deegan et al. 1997). Indicator-based SP valuation combines the use of ecological indicators within SP surveys with formal ecological models linking those indicators to assessment endpoints from which utility is derived (Johnston et al. 2010). The model—shown conceptually by figure 1 above—allows the use of easily-understandable indicators within survey scenarios (the economic component), yet provides unambiguous ecological linkages among these indicators and the assessment endpoints that determine values (the ecological component). As a result, the approach generates WTP measures that may be formally and unambiguously linked to ecological models of ecosystem function and ecological risk assessments, are based on measurable policy outcomes, and as a result may be more easily incorporated into policy-based benefit cost analysis.

Despite the promise of such methods, development of indicator-based valuation methods is complicated by a number of factors. First, assessment endpoints relevant to the public may not match those of greatest relevance to scientists. Hence, the first challenge is to communicate ecological linkages implicit in existing indices to non-scientist survey respondents—such that they may appropriately use this information to provide useful inferences regarding valued aspects of marine ecosystems. Second, it is necessary to determine those direct or indirect indicators—and the form of those indicators—that most successfully communicate baselines or changes in assessment endpoints. As shown by Schiller et al. (2001), Scruggs (1997) and others, technical measurements of greatest relevance to scientists may provide little meaningful information to non-scientists. This problem is compounded when survey and respondent constraints limit the amount and detail of information that can be included and processed (Johnston et al. 2004a). However, multi-metric biological indicators have been shown to perform well in some revealed preference models (USEPA 2000) and SP models (Johnston et al. 2010). Hence, one of the main issues here is communication of the technical measures to survey respondents.

When presenting technical single- or multi-metric indicators to survey respondents, the choice of units and communication of indicators is critical. Even within the ecological literature, the choice of units has been the subject of considerable controversy, with some suggesting that multi-metric indicators such as the IBI (indices of biological integrity) can produce "nonsense" results based on ambiguous measurement units (Suter 1993). Given such criticism from scientists, it would seem unlikely that non-scientists—including survey respondents—would have the capacity to usefully interpret such scales. In contrast, Schiller et al. (2001) proposes and qualitatively tests "common language indicators" designed to represent multi-metric measures in easily understood form. For valuation purposes, the challenge in such cases is to be able to link common language indicators back to quantifiable assessment endpoints—and whether such indicators provide sufficient information on quantifiable ecosystem characteristics to permit meaningful value elicitation, including sensitivity to scope (Smith and Osborne 1996; Veisten et al. 2004). Moreover, ambiguities with underlying multi-metric indicators will carry through to associated common language indicators.

A STANDARDIZED BASIS FOR VALUING ECOSYSTEM BENEFITS & BENEFIT TRANSFER

Methods for the valuation of ecosystem benefits are extensively described in the ecological economic literature and are generally an extension of the economic valuation approach used in cost-benefit analysis (see, for general overviews, Barbier 2007; Chee 2004; De Groot et al. 2002; King and Mazzotta 2000; Nijkamp et al. 2008). Use of these methods in empirical studies of ecosystem services has tended to be more limited, applied in most cases to global or regional scale and focusing on total rather than marginal values (see Costanza et al. 1997; Farber et al. 2006).

Similarly, explicit empirical valuation of ecosystem services & benefits associated with MPAs are rare (see Klein et al. 2008; Mardle et al. 2004; Roncin et al. 2008) among a more extensive, albeit still limited literature, on empirically derived economic costs and benefits of MPAs (see for example Alban et al. 2008; Carter 2003; Hoagland et al. 1995; Rudd 2007). Barbier (2007) used production function analysis to value ecosystem services from mangrove ecosystems in Thailand. Mardle et al. (2004) studied the insurance value of MPAs in Sicily, and Sumaila (2002) carried out a similar study for cod in the North Sea.

Carrying out site-specific studies using any of the above techniques is costly and time consuming. In many cases, each method provides only a partial valuation of ecosystem benefits (Costanza 2008; Plummer 2009). Because of this, benefit or value transfer is an increasingly used approach to value environmental goods and services (Iovanna and Griffiths 2006; Plummer 2009; Wilson and Hoehn 2006). Benefit transfer is the practice of applying estimates of economic value from one location to a similar site in another location (Plummer 2009).

Despite the mixed performance of benefit transfer in past assessments (Rosenberger and Stanley 2006; Smith et al. 2002), welfare measures estimated using such methods are increasingly incorporated as central components of benefit cost analyses (Bergstrom and De Civita 1999; Griffiths and Wheeler 2005; Iovanna and Griffiths 2006; Johnston and Rosenberger 2010). Given the generally unreliable performance of unadjusted single-site transfers, however, researchers are increasingly considering approaches that allow welfare measures to be adjusted for characteristics of the policy context under consideration (Rolfe and Bennett 2006; Rosenberger and Loomis 2000; Rosenberger and Phipps 2007; Smith et al. 2002). The validity of WTP adjustments and their appropriateness for benefit transfer depends on the presence of systematic and identifiable variation in underlying WTP (Desvousges et al. 1998; Johnston et al. 2003; Brouwer 2002; Rosenberger and Stanley 2006). If WTP cannot be shown to vary systematically according to the principal attributes distinguishing study and policy sites, the justification for such adjustments becomes more tenuous.

One of the major obstacles to the development of transferable benefit functions is the difficulty in synthesizing and comparing information from available valuation studies. This difficulty is compounded by the lack of standard representations of measurable ecological changes in existing studies (Johnston et al. 2003; Loomis and Rosenberger 2006). The development of standardized ecological indicators for SP valuation of marine resources provides a means to alleviate problems that arise in comparing valuation estimates due to incompatibility of environmental quality measures.

THE ROLE OF STAKEHOLDERS AND USE OF DELIBERATIVE DISCOURSE

Deliberative discourse-based evaluation techniques can be used where small groups of local community members, agency representatives and stakeholders are involved in the validation and/or modification of both the benefit transfer estimates and the qualitative assessment of benefits that have not been subject to economic valuation (Kumar and Kumar 2008; Spash 2008; Wilson and Howarth 2002). Many ecological benefits and costs are not easily valued in monetary terms and monetization as a process can disenfranchise the very communities most impacted by changes in ecosystem services and ecosystem benefits (Cowling et al. 2008; Wilson and Howarth 2002).

Engaging stakeholders in the identification and assessment of ecosystem services benefits uses group deliberation to inform knowledge about ecosystem services, societal relationships to ecosystem benefits and the value of benefits (Cowling et al. 2008). It also has wider benefits including: (1) Understanding potential for conflict over multiple objectives for spatial management of coastal ecosystems; (2) Better specification of existing interactions between ecosystems services and the communities that depend on them and; (3) Disseminating knowledge about ecosystem services and benefits to coastal communities, decision makers and stakeholders; fostering community participation in marine spatial planning (Kumar and Kumar 2008; Lynam et al. 2007; Pomeroy and Douvere 2008).

Following Schiller et al. (2001), focus groups and individual interviews can be used to assess perceptions among the public about marine life and marine ecosystem condition and ecosystem services, and to assess linkages between representations made by indicators and underlying preferences and perceptions of non-expert individuals (Johnston et al. 1995). Particular emphasis could be given to issues often suppressed in SP valuation (e.g., uncertainty and the relevance of resource composition and diversity). Criteria for the selection of indicators includes correspondence to ecological models, appropriateness for economic valuation, and information that is meaningful, comprehensive, and comprehensible to non-scientists (Schiller et al. 2001).

EVALUATING TRADE-OFFS BETWEEN ECOSYSTEM SERVICES AND BENEFITS

There may be hundreds of initial ecosystem services to consider when engaged in MPA planning, but only a relative few are critical for determining any given issue. Recognizing this our proposed evaluation framework builds on the preceding indicator-based evaluation and deliberative discourse components to explore ecosystem services and benefits using a decision analytical structure. Typically, decision makers receive generalized types of information such as the outputs from modeling and monitoring studies, risk assessment, cost or cost-benefit analysis, and stakeholder preference surveys. However, the associated assessment and decision process is often ad hoc with little structure and guidance on how to integrate or judge the relative importance of information from each source (Kiker et al. 2005). A structured decision-support approach guides the integration of different data sources, including stakeholder values and stated preferences. It provides guidance on which information is the most useful for MPA planning and where further research and monitoring efforts should be directed.

Multi-criteria assessment (MCA) methods are structure decision support approaches oriented to the multi-dimensional character of many natural resource management problems (Hajkowicz 2007; Kiker et al. 2005). They are designed to overcome, when assessing trade-offs, the problems of multiple objectives, incommensurate units, the need to consider both qualitative and quantitative data and the need to incorporate stakeholder knowledge and preferences (Chee 2004). These tools are inherently capable of integrating biological, social and economic data. They are ideal for assisting evaluation in data-poor situations such as ecosystem services (Fisher et al. 2009). Brown et al. (2001) used MCA to complete a trade-off analysis for a marine protected area in Tobago. The researchers found that the MCA-based trade-off analysis enhanced stakeholder participation and understanding of the issues associated with MPA planning.

Many kinds of MCA approaches are documented in the decision support and evaluation literature (Kiker et al. 2005). Each attempts to assess tradeoffs between alternative management actions or scenarios against a set of evaluation criteria (Kiker et al. 2005). Belton and Steward (2002) document the strengths and weaknesses of the main approaches. Recently, MCA methods have been developed to include Bayesian algorithms (Ullman, 2006). Bayesian based approaches are especially useful when decision making is characterized by uncertainty where the available information is imprecise and incomplete (Chee 2004)

Even with an indicator-based evaluation framework, much of the information available about ecosystem services and benefits will be uncertain, incomplete and evolving. Information may be conflicting and subject to different interpretations. Some of it may be based on good science, some on limited science and some on opinion. Some of it may be qualitative and some quantitative, some spatial and some not. Regardless of all this messiness, all must be integrated with uncertainty modeled from the beginning, not simply addressed at the end of the planning process through the invocation of the precautionary principle. Ex-post treatment of uncertainty is at best a blunt approach to a complex issue and at worst a non-transparent undermining of the decision making process.

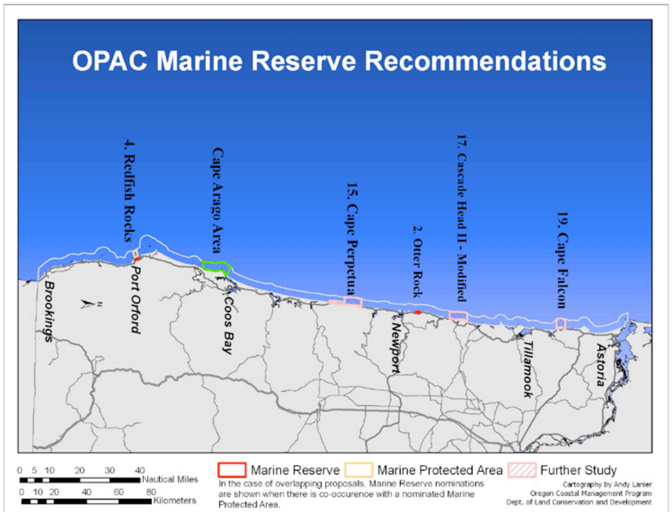
As well as endogenously incorporating uncertainty, a particular strength of the Bayesian MCA algorithms described by Ullman (2006) is the value of information (VOI) function. This is a decision analytic technique that explicitly evaluates the benefit of collecting additional information to reduce or eliminate uncertainty. Ullman (2006) further describes a what-to-do-next analysis. This is an expert system that uses the VOI results, combined with the level of consensus, demonstrated knowledge and assessed ecosystem benefits to generate statements that guide groups to the most cost-effective actions required to address unresolved albeit important issues.

Embedding a structured decision analytic within our proposed framework creates a powerful heuristic tool for integrating ecosystem services and benefits and exploring tradeoffs between them. Stated preference values, other un-monetized information, together with agency, community and stakeholder input can then be used to develop recommendations for research that generates the greatest values of information, alternative MPA boundaries, monitoring of expected outcomes and guiding adaptive management programs.

PROPOSED APPLICATION TO MARINE RESERVE PLANNING IN OREGON, USA

In late 2008, Oregon's Ocean Policy Advisory Council (OPAC), following a charge from Governor Kulongoski, reported back on the creation of a series of marine reserves along the Oregon coast (OPAC 2008). OPAC nominated two pilot marine reserve sites: Redfish Rocks in Port Orford and Otter Rock in Depoe Bay (OPAC 2008). They also nominated four other possible sites for further study and consultation. The two pilot and four potential sites are shown in Figure 2. In early 2009, Governor Kulongoski responded to OPAC's recommendations and recommended the State Legislature set aside funds for the establishment of the two pilot reserves and the further evaluation of the other four sites.

Figure 2: Proposed Oregon Marine Reserve Locations



Our working hypothesis is that the evaluation framework for ecosystem services described in preceding sections can be applied to the planning, assessment and monitoring of marine reserves in Oregon.

We will test this hypothesis in an interactive manner as illustrated in Figure 3. First we will characterize the ecosystem services at the proposed marine reserve sites and derives candidate indicators of expected long-term change in ecosystem services that are expected as a result of creating the reserves. These indicators represent a convergence between ecological models and the understanding and cognitive abilities of stakeholders and Oregon residents. This phase includes development of structural linkages between previously developed indicators or indices of change and assessment endpoints that influence individual and community wellbeing, together with qualitative research to develop related indicators readily understandable by non-scientists. The goals of this initial phase are to identify an initial set of candidate indicators to serve as a starting point for survey development and to develop ecological-economic models allowing use of indicators for SP valuation.

Of particular focus will be the consolidation or reduction of ecological indicators—often involving multiple dimensions or metrics—into a smaller number of metrics or descriptive attributes suitable for survey use. It is critical that in transforming ecological indicators into simpler forms, the underlying information is not overly diluted, such that indictors remain consistent and meaningful, and resulting survey indicators remain unambiguously linked to the underlying ecological parent indicators based on ecological models. Additional considerations relate to the role of uncertainty. Preference will be given to indicators that explicitly reflect uncertainty implicit in the forecasting of short- and long-term impacts on marine resources.

Simultaneous to the development of an initial set of ecological indicators that satisfy minimum theoretical criteria, we will conduct qualitative research to assess the suitability of various indicators with regard to public perception and cognition. This will include focus groups and individual interviews to assess perceptions among the public about aquatic life and marine ecosystem condition, and to assess linkages between representations made by indicators and underlying preferences and perceptions of non-expert individuals. Particular emphasis will be given to issues often neglected in economic valuation (e.g., uncertainty and the relevance of resource composition and diversity). Criteria for indicators at this stage will include correspondence to ecological models, appropriateness for economic valuation, and information that is meaningful, comprehensive, and comprehensible to non-scientists.

Having identified candidate indicators, we will next identify how these indicators are expected to change as a result of the creation of the marine reserve. This will be based recently completed reviews of changes in the composition, structure, function of marine reserves in temperate regions of the world. Using deliberative discourse methods to gain stakeholder and expert input we will parametrize the likely magnitude of the changes in these indicators and the uncertainty associated with these changes/

Next will explore likely tradeoffs in ecosystem services and benefits associated with the creation of MPAs using a decision analytical structure. Typically, decision makers receive generalized types of information such as the outputs from modeling and monitoring studies, risk assessment, cost or cost-benefit analysis, and stated preference surveys. However, the associated assessment and decision process is often ad hoc with little structure and guidance on how to integrate or judge the relative importance of information from each source, MCA forms the basis for this a structured decision-support approach, guiding the integration of different data sources, including stakeholder values and stated preferences. What to do next and VOI functions of the MCA will provide guidance on which information is the most useful to MPA planning in Oregon generally and where further ecological and economic research and monitoring efforts should be directed to assess the effectiveness of the MPAs over time.

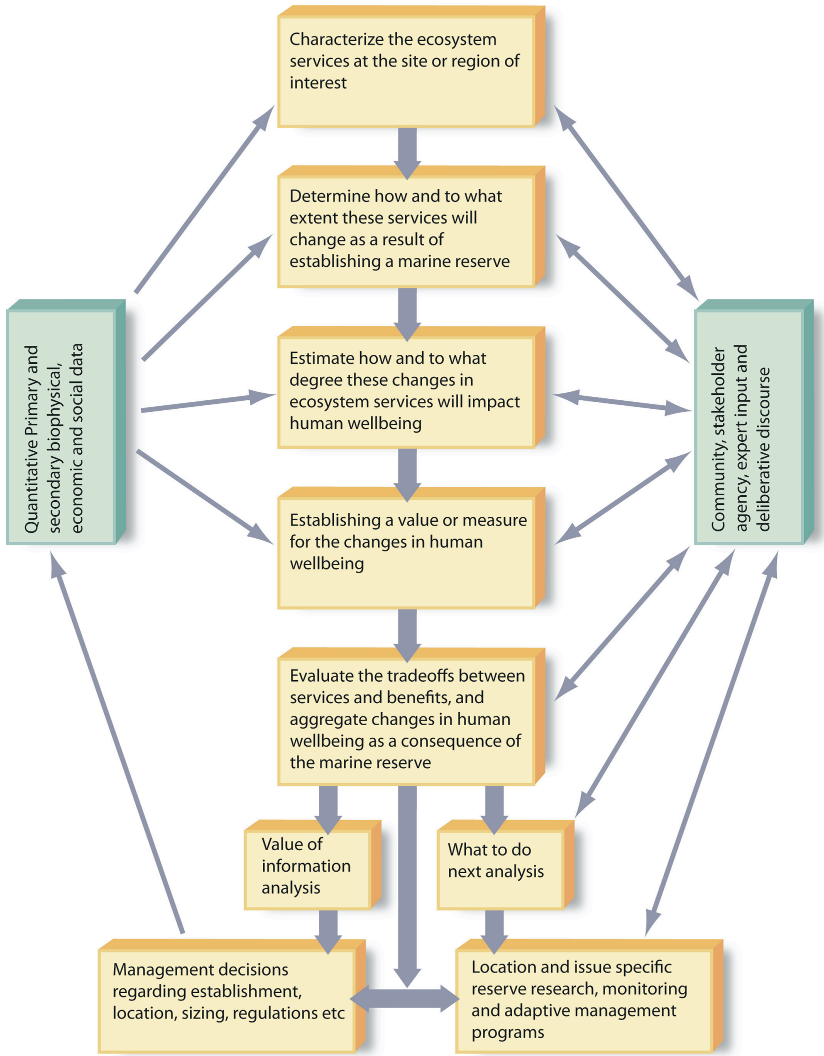


Figure 3: MPA Assessment Framework

SUMMARY

Our framework first integrates economic and ecological models to develop indicators to communicate baselines and changes in marine ecosystem services and benefits. These indicators provide the foundation for valuation of ecosystem benefits using stated preference surveys. These indicators, stated preference information and related models linking indicators to management endpoints can then be integrated within a decision analytical approach to assess tradeoffs associated with the design and implementation of MPAs. Combined with stakeholder involvement formalized through deliberative discourse method, indicator-based methods should improve the validity and accuracy of use/nonuse benefit estimation for marine ecosystem services and benefits. The integrated economic/ecological models also link measurable policy outcomes to welfare-relevant ecosystem services. Once applied, our framework may demonstrate that indicators of ecosystem services and benefits can unambiguously communicate applicable ecological baselines and marginal changes in these baselines resulting from marine resource management actions such as the creation of MPAs.

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