Levin, P. S., G. D. Williams, A. Rehr, K. C. Norman, and C. J. Harvey. 2015. Developing conservation targets in social-ecological systems. *Ecology and Society* **20**(4):6. http://dx.doi.org/10.5751/ES-07866-200406

Insight

Developing conservation targets in social-ecological systems

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ABSTRACT. The development of targets is foundational in conservation. Although progress has been made in setting targets, the diverse linkages among ecological and social components make target setting for coupled social-ecological systems extremely challenging. Developing integrated social-ecological targets is difficult because it forces policy makers to consider how management actions propagate throughout social-ecological systems, and because ultimately it is society, not scientists, that defines targets. We developed an interdisciplinary approach for identifying management targets and illustrate this approach using an example motivated by Puget Sound, USA. Our approach blends ecological modeling with empirical social science to articulate trade-offs and reveal societal preferences for different social-ecological states. The framework aims to place information in the hands of decision makers and promote discussion in the appropriate forums. Our ultimate objective is to encourage the informed participation of citizens in the development of social-ecological targets that reflect their values while also protecting key ecosystem attributes.

Key Words: *conservation target, ecosystem assessment, scenario analysis, social norm analysis,*

INTRODUCTION

How much conservation is enough? How much area should we set aside to safeguard the ecological processes that support ecosystem function or protect the viability of native species? What are the social or economic costs associated with achieving biodiversity objectives? Such questions are foundational in conservation and environmental management, but remain a vexing problem on the ground because they force practitioners to confront the social and economic constraints inherent in any conservation endeavor. This is particularly true in habitats near human population centers or human activities, where trade-offs are potentially more acute.

The challenge of developing conservation targets, i.e., explicit goals that quantify the amount of an ecosystem component to be conserved, is one of the major tasks of modern conservation (Carwardine et al. 2009) and has received considerable attention from natural and social scientists (e.g., Byers et al. 2001, Watson 2003, Tear et al. 2005, Sanderson 2006, Geisler 2010, Di Minin and Moilanen 2012). In many single-species and single-sector management arenas, targets are well established. For example, the U.S. Endangered Species Act requires objective, measurable criteria that, when met, would result in the delisting of an endangered or threatened species; the Magnuson-Stevens Sustainable Fisheries Act mandates the development of fishing harvest targets and other reference points that are related to maximum sustainable yield, modified by ecological and economic considerations (Levin 2014); and the European Union's Water Framework Directive set the goal of achieving "good status" for all of Europe's surface waters (Hering et al. 2010).

Considerable progress has been made in setting targets for specific components of ecosystems (e.g., Moilanen and Arponen 2011). And, as ecosystem-based management (EBM) becomes the norm, the urgency to develop targets for entire coupled social-ecological systems has increased (Samhouri et al. 2012). For instance, the notion of optimum yield in fisheries, i.e., the yield that maximizes fisheries benefits across ecological, economic, and social domains, is a holistic target that is receiving increased attention (Levin 2014). Because of the many and diverse linkages among ecological and social components of ecosystems, the challenges of developing meaningful system-scale targets are amplified. In part, this difficulty arises because, as Lackey (1998) noted, a key objective of EBM is to maintain ecosystems of sufficient condition that they provide desired social benefits. Critically, however, it is society (or powerful parts thereof), not scientists, that define desired social benefits. Thus, any consideration of targets must incorporate not just ecological understanding, but also dynamic societal values.

Developing integrated social-ecological targets is challenging because it compels policy makers to explicitly consider how management actions propagate throughout human and biophysical domains of ecosystems. For example, the articulation of a management target for habitat restoration must be linked to targets for species that depend on that habitat; thus, habitat targets may promulgate throughout the ecological community, influencing targets for a wide range of ecosystem components and species that depend on specific habitats (Kaplan et al. 2012, Plummer et al. 2013). Some targets may be bounded by regulatory or legislative mandates. However, societal tolerance for monetary costs such as taxes and user fees or nonmonetary costs such as changing development/transportation patterns, including cultural costs (Chan et al. 2012*a*) associated with specific targets, may also constrain the political feasibility of some targets (Naidoo et al. 2006, Hicks et al. 2009).

We developed a framework for identifying management targets in social-ecological systems, and we illustrate this approach using an example motivated by seagrass restoration in Puget Sound, USA. In doing so, we highlight a means to blend ecological and social science to inform the creation of scientifically rigorous and socially responsive targets.

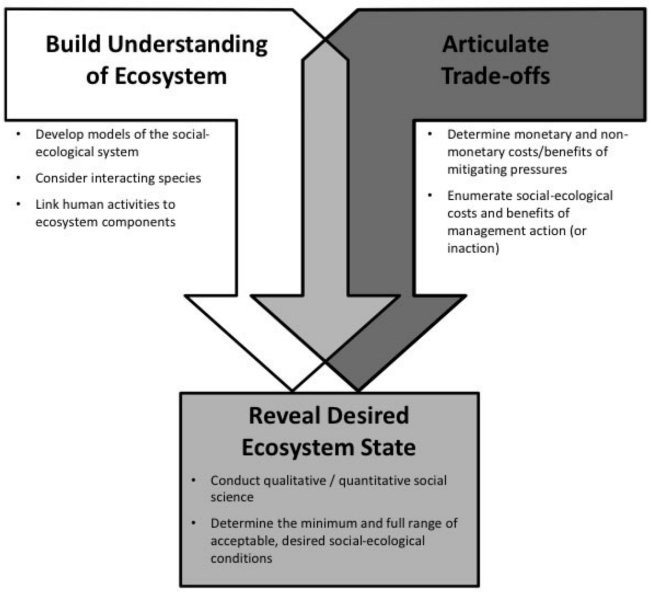
A GENERAL FRAMEWORK FOR DEVELOPING SOCIAL-ECOLOGICAL MANAGEMENT TARGETS

Although a number of methods exist for developing EBM targets (e.g., Pressey et al. 2003, Samhouri et al. 2010, 2012, Large et al.

[1]NOAA Fisheries, Northwest Fisheries Science Center, [2]Pacific States Marine Fisheries Commission, [3]Northwest Fisheries Science Center, NMFS, NOAA, [4]National Marine Fisheries Service

2013), these approaches suffer from two major issues: (1) they consider one species or ecosystem indicator at a time and disregard linkages among ecosystem components and (2) they do not consider societal preferences. To overcome these shortcomings, we propose the following general framework (Fig. 1).

Fig. 1. A three-step approach for developing conservation targets for social-ecological systems. Developing targets begins by building an understanding of the system through the use of conceptual and numerical models. It continues by articulating trade-offs that highlight the costs and benefits of different management scenarios or states of nature. Finally, it uses appropriate tools of social science to determine the preferred state of the system, given the trade-offs inherent in any management option.



Build an understanding of the social-ecological system

Because social-ecological systems have interacting components, developing targets for any individual element of the system requires understanding how it is linked to other salient ecosystem components. Developing an understanding of the social-ecological system, including the biophysical, socioeconomic, and management systems that affect the ability to achieve management goals, is the first step in in this framework. Conceptual ecosystem models have proven useful for understanding ecosystems and synthesizing diverse scientific information (Ogden et al. 2005). Conceptual models have proven useful in improving outcomes in resource management, especially when scientists, resource managers, and stakeholders jointly develop models (Svarstad et al. 2008, Chan et al. 2012*b*).

Numerical models that can simulate changes to biophysical or socioeconomic components of the system are most useful. This may be the application of relatively simple approaches such as fuzzy-logic cognitive maps (Gray et al. 2012), loop analysis (Carey et al. 2013), or Bayesian Networks (Uusitalo 2007) that build upon conceptual models. In other cases, where quantitative capacity exists, complex simulation models such as Ecopath with Ecosim (EwE; Christensen and Walters 2004) or Atlantis (Fulton et al. 2011) can be applied. The choice of model will depend on the particular scientific and management arena (Plagányi 2007); however, no matter what modeling approach is used, the aim is to understand how changes in focal ecosystem components and human activities propagate through the system as well as the uncertainty of these connections.

Articulate trade-offs

Trade-offs are at the heart of EBM. Most commonly, researchers and managers consider trade-offs among potentially competing objectives such as those related to resource extraction, energy production, or conservation (Levin et al. 2009). Individuals differ in the values they hold (Rockeach 2008); therefore, they will differ in how they assess trade-offs (Hicks et al. 2009). Costs of implementing various management options can vary greatly and certainly have the potential of influencing the nature of trade-offs. Importantly, implementation costs are not only monetary; a number of nonmonetary costs related to culture, transportation, and livability are associated with management actions and indeed have been the subject of intense research (e.g., Chan et al. 2012*a*, Tallis et al. 2012). Consequently, in addition to examining trade-offs among management objectives, it is important to determine the economic, social, and cultural costs of different management options (Krutilla 1967, Nelson 2006).

Assess ecosystem preferences

Economists and social scientists have developed numerous approaches for evaluating the desirability of an environmental state to individuals or communities (Haab and McConnell 2002, Nelson 2006, Plummer 2009, Guerry et al. 2012). Economists, for example, have methods for estimating values of ecosystem components that cannot readily be bought or sold (e.g., Carson et al. 2001). Social scientists have used quantitative analysis of communication content (Neuendorf 2002), systematic surveys (Safford et al. 2014), and workshops and individual interviews (Donatuto et al. 2014) to evaluate preferences. Regardless of approach, the critical aspect of this step is that desirability is assessed for the complete suite of components of the social-ecological system. Preferences are thereby revealed in a manner that accounts for linkages, costs, and benefits inherent in the system. Knowing the desired social-ecological system state, the minimum acceptable state, and range of acceptable conditions, policy makers can develop management targets that reflect what is possible as well as what citizens want.

An important aspect of this framework is that it can be bounded by ethical or social norms such that targets can only occur within acceptable levels. In some regions, such norms will be codified by legislation or regulations regarding things such as clean water and air, human health, endangered species, sustainable harvests, and so forth. In other instances, norms may not be codified, necessitating additional social science research. Thus, this approach to target selection serves to identify what social-ecological states are desirable within the area society has already deemed generally acceptable.

DEVELOPING SOCIAL-ECOLOGICAL TARGETS: A PUGET SOUND–INSPIRED CASE STUDY

We illustrate our social-ecological approach for setting targets with a case study inspired by Puget Sound, USA. This case study is stylistic and simplified but motivated by real-world issues confronting this region. The Puget Sound ecosystem includes 41,500 km² of marine, freshwater, and upland habitats in a watershed that supports large and growing population centers around the Seattle and Tacoma region. The area also contains rural communities that have close ties to forestry, fishing, and agriculture. Today, Puget Sound is also home to 19 Native American tribes, the Southern Coast Salish, and their culture and communities are tightly linked to the Sound (Thrush 2009). The waters and shorelines of Puget Sound make up a unique estuarine system that is valued for its beauty and ecological importance (Safford et al. 2014). Nonetheless, like other marine and estuarine ecosystems, Puget Sound has been affected by a variety of human activities, including agriculture, heavy industry, timber harvest, fishing, and the development of sea ports and residential property. As a consequence, the Puget Sound ecosystem is degraded, and the processes supporting it are impaired.

For this case study, we focused on social-ecological targets for eelgrass, a habitat that is highly valued for its ecological and economic benefits (Mumford 2006, Plummer et al. 2013). Its canopy forms three-dimensional complexity that provides vital spawning, foraging, nursery, and settlement habitat for many species (Mumford 2006). As examples, commercially, recreationally, and culturally important juvenile Pacific salmon (genus *Oncorhynchus*) and Dungeness crabs (*Carcinus magister*) use eelgrass as a refuge from predation; it provides important spawning substrate for Pacific herring (*Clupea pallasii*), a key prey species; and a number of fish species forage in and around eelgrass beds. Beyond these direct effects, changes in eelgrass also have indirect effects that permeate through nearshore food webs. For example, increases in eelgrass are associated with increased abundance of predators valued by Puget Sound residents, such as bald eagles (*Haliaeetus leucocephalus*), harbor seals (*Phoca vitulina*), and sea lions (*Zalophus californianus*). The increase of these predators is subsequently associated with changes in the Puget Sound prey community (Plummer et al. 2013). Additionally, the Coast Salish people used eelgrass in ceremonies, and the collection and processing of eelgrass were central in the transfer of traditions and knowledge across generations.

Currently eelgrass covers about 22,800 hectares in Puget Sound, mostly in large shallow embayments (Essington et al. 2011). Long-term trends in eelgrass abundance are uncertain; even so, there is some concern about loss of seagrass, particularly in smaller, isolated sites (Essington et al. 2011). Eelgrass in Puget Sound is threatened by a number of activities that reduce light or disturb the sediment in nearshore regions, including shoreline armoring, overwater structures such as marinas and docks, and nutrient and sediment loading (Rehr et al. 2014*a*).

BUILDING AN UNDERSTANDING OF THE PUGET SOUND FOOD WEB

For this work, we used EwE, a modeling approach that simulates the trophic ecology of interacting functional groups. The core EwE equations are documented elsewhere (Christensen et al. 2005), as are details of the Puget Sound parameterization of EwE (Harvey et al. 2012). Following Plummer et al. (2013), we added “mediating effects” to the model, whereby appropriate predator-prey relationships were influenced by the abundance of eelgrass. Specifically, eelgrass could either lower the foraging efficiency of a predator, e.g., by providing refuge habitat for prey, or enhance a predator’s foraging efficiency, e.g., by aggregating prey and rendering them more vulnerable. Changing eelgrass biomass affects the rest of the model food web via three pathways: the relative availability of eelgrass for grazers, e.g., waterfowl; the amount of dead eelgrass that contributes organic matter to the detrital pool; and by mediating ecological interactions, e.g., by altering the strength of predator-prey relationships.

BUILDING AN UNDERSTANDING OF HUMAN ACTIVITEIS THAT MODIFY PUGET SOUND EELGRASS

Because data regarding the functional relationship between human activities and Puget Sound eelgrass were lacking, we conducted an expert elicitation in which experts were asked how they would expect eelgrass cover to respond to incremental changes in nutrient loading, sediment transport, overwater structures, and shoreline armoring (Rehr et al. 2014*a*). Rehr and colleagues (2014*a*) then used a Bayesian network analysis to generate the relationships between changing human activities and eelgrass. Coarse estimates of costs of mitigating these impacts, as well as potential impacts on commute time and property values associated with this mitigation (e.g., modifying storm water runoff or reducing shoreline armoring), were provided by staff at the Puget Sound Partnership (http://www.psp.wa.gov), the state agency charged with overseeing Puget Sound recovery. Our research thus focused on costs associated with specific management actions as perceived by government agency staff. As such, our treatment does not explicitly examine all potential social and cultural impacts (e.g., Poe et al. 2014) associated with seagrass restoration.

ARTICULATING TRADE-OFFS

We examined trade-offs among 16 metrics. We selected seven biological indicators (biomass of adult herring, wild Pacific salmon, bald eagles, herbivorous birds, gulls (genus *Larus*), southern resident orca whales [*Orcinus orca*], and eelgrass) from the EwE model output because of their salience in the Puget Sound management community (e.g., http://www.psp.wa.gov/vitalsigns/index.php). Four metrics were related to human stressors: area of impervious surfaces, nutrient loading, number of overwater structures, and amount of shoreline armored. Two parameters, rural growth and urban density, were selected to represent patterns of development. Finally, property value, direct costs of restoration or management, and commute time were selected as indicators of costs.

We explored trade-offs among these variables in seven scenarios (Table 1). These scenarios were based loosely on the “unconstrained” and “managed growth” alternative futures analysis of Bolte and Vache (2010), which provided us with projected changes in sediment and nutrient inputs, shoreline armoring, and overwater structures. Other aspects of the scenarios were guided by growth management, transportation, and land use patterns outlined by the Puget Sound Regional Council (http://www.psrc.org/data), and related habitat protection and restoration targets and thresholds set by the Puget Sound Partnership. Using the functional relationships between the stressors and eelgrass from Rehr et al. (2014*a*), we approximated the change in eelgrass associated with each scenario.

Table 1. Description of scenarios examined in the Puget Sound–inspired case study.

Scenario Land Use / Growth	Shoreline Development or Restoration	Approximate % change in eelgrass
1. Status Quo (SQ)		
Assumes SQ 2060 levels of impervious surface area (7%; Bolte and Vache 2010). Development pattern emphasizes moderate density uses. Some conversion of undeveloped lands to both commercial, residential, and park uses is allowed. Areas within an urban growth area, near roads, or with a water view are more likely to be developed (Bolte and Vache 2010). Uses 2010 estimates of urban density/housing (single family 62.3%, multifamily 32.8%, mobile home/other 4.9%), transportation (69.5% drive alone, 11.2% carpool, 8.6% public transportation, 3.5% walk, 0.9% bike, 1.1% other, 5.2% work from home), and commute times (27.6 min transit time; derived from http://www.psrc.org/data).	No development is allowed on deltas, within floodplains, or in areas with unstable slopes; development on existing wetlands is limited. Areas containing significant wetlands are less likely to be developed. Maintains a moderate level of protection of wetlands, some restoration of historic wetlands; a moderate level of protection of existing open space areas, and a moderate level of protection of areas adjacent to eelgrass beds and herring spawning areas (derived from Bolte and Vache 2010).	0%
2. Unconstrained Growth II and Double population		
Similar to Scenario 3, but with doubling of the projected population. Assumes use of single occupant vehicles and commute times doubles relative to Unconstrained Growth I		-35%
3. Unconstrained Growth II		
	Similar to Scenario 4, but uses projected increases of 20% in shoreline armor and overwater structure based on current rates of development (e.g., 30 new overwater structures and 6.4 km shoreline armoring/y).	-25%
4. Unconstrained Growth I		
Development pattern emphasizes low-density uses. Includes 40% increase in impervious surface area relative to SQ2060 (derived from Bolte and Vache 2010). Assumes increased use of single occupant vehicles, and higher average commute times (derived from http://www.psrc.org/data).	Reflects a relaxation of land-use restrictions with limited protection of ecosystem functions. The Unconstrained Growth I scenario allows significant new development in the nearshore. No development is allowed on deltas or on unstable slopes, but other shoreforms are developable. Includes 5% percent increase in the amount and density of nearshore modifications (shoreline armor and overwater structures) relative to SQ2060 (derived from Bolte and Vache 2010).	-15%
5. Managed Growth I		
Reflects the adoption of an aggressive set of land-use management policies focusing on protecting and restoring ecosystem function and concentrating growth within Urban Growth Areas and near regional growth centers. In existing developed areas, focus is on increasing density. Creation of parks in developed areas is included. Existing open space is precluded from development. Includes a 20% reduction in impervious surface area relative to SQ2060 (derived from Bolte and Vache 2010). Assumes increases in urban density/multifamily housing, higher use of mass transit, and lower average commute times (derived from http://www.psrc.org/data).	No new development is allowed within 200 m of the shoreline. Outside the 200 m zone, development is severely restricted in areas near sensitive lands, including current and historic wetlands, lands with significant conservation opportunities, or lands adjacent to streams. Water views are protected from development. This scenario reflects a high level of protection of existing and undeveloped historic wetlands; aggressive restoration of historic wetlands, and protection of sites with high conservation/restoration potential. No development is allowed next to eelgrass/Pacific herring (*Clupea pallasii*) spawning areas. Includes a 5% reduction in the amount and densities of nearshore modifications (shoreline armor and overwater structures) relative to SQ2060 (derived from Bolte and Vache 2010).	+10%
6. Managed Growth II		
	Similar to Scenario 5, but reflects the adoption of Puget Sound Partnership ecosystem recovery and related targets and some proposed targets articulated in the Washington Shoreline Management Act (20% reduction in shoreline armoring and overwater structures, based on upper limit of preliminary proposals at the time). Local Shoreline Master Programs meet an overall standard of No Net Loss for ecological function. The total amount of armoring removed is greater than the total amount of new armoring; feeder bluffs receive strategic attention for removal of existing armoring and avoidance of new armoring.	+25%
7. Unpopulated		
A scenario in which the Puget Sound region is unpopulated and no development exists.		+45%

We next evaluated the outcomes of the seven scenarios using the Puget Sound EwE model, where eelgrass abundance influenced the community directly and indirectly as previously described. The model was run for 50 years following each eelgrass perturbation to allow the system to reach a new equilibrium state.

REVEAL DESIRED ECOSYSTEM STATE

The normative approach is a powerful way to collect and organize data about stakeholder values (Vaske et al. 1993). Norms define what is considered normal or generally accepted within a cultural context, and may serve as societal standards to evaluate ecosystem conditions, human activities, or management strategies. Norms are typically described by a graphic device called a social norm curve (Jackson 1965); in our case, the x-axis represents social-ecological state and the y-axis portrays stakeholder preferences.

To generate social norm curves, we queried 128 people who were drawn from interested, voluntary audiences identified by querying key informants representing major stakeholder groups in the region, including environmentalists, shellfish aquaculture, recreational fishing, coastal development, and government. Participants ranged in age from 18 to 84 years old (mean age, 51 years), and were 40% female, 60% male. Among the participants, 45% identified themselves as Democrats, 15% as Republicans, and 40% as Independents, and all were residents in the Puget Sound region. Once identified, participants were assessed to ensure that the range of included individuals corresponded to the range of environmental awareness of Puget Sound residents generated by Safford et al. (2014).

Participants were exposed to the scenarios in two ways. We first used radar plots (Fig. 2), which have proven useful for visualizing multidimensional trade-offs (e.g., Guerry et al. 2012). However, because the numerical nature of radar plots may not resonate with some respondents (Bell 1984), we also developed stylized images (Fig. 3). Such images offer an effective alternative for visualizing complex systems and informing environmental policy (Fiore et al. 2009), and have previously been used for illustrating different states of nature in normative surveys (Bateman 2009). We vetted our images to ensure that they effectively communicated the numerical data underlying the images, and details are provided in the article by Rehr et al. (2014*b*). Briefly, we engaged a digital production agency (Studio 216, http://www.studio216.com) to generate computer visualizations that were quantitatively linked to the scenarios described above and in the article by Bolte and Vache (2010). We then pilot-tested the computer-generated images to determine the degree to which individuals could resolve differences in the visualizations. Rehr and colleagues (2014*b*) showed that coarse differences of the sort we used in this study were distinguishable by participants.

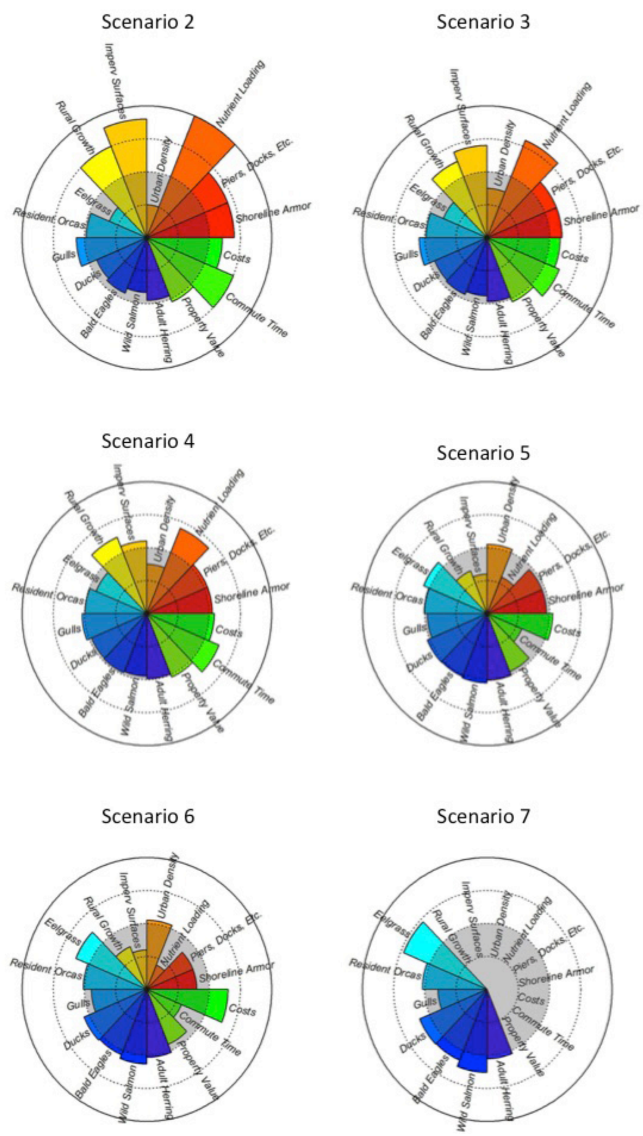
A trained, professional facilitator presented both the radar plots and the computer-generated images to participants. The facilitator systematically explained the key features of each graphic. We then asked participants to score the desirability of each scenario on a Likert scale from –2 (completely unacceptable) to +2 (optimal state). By requiring a single score for each scenario, we effectively forced participants to consider the trade-offs inherent in any management action or inaction.

Fig. 2. Examples of the visualizations used to examine the desired state of the social-ecological system. Depicted are an (A) overview, (B) urban center, (C) outlying region (rural growth and open space), (D) shoreline, and (E) subtidal marine environment for a stylized Puget Sound metropolis. Two scenarios are illustrated: scenario 2 in which growth is unconstrained and population rapidly grows, and scenario 5 in which growth is managed (see Table 1 for details).

CASE STUDY RESULTS

Our scenarios depicted a range of development schemes in which rural growth, impervious surfaces, shoreline armor, overwater structures, and nutrient loading were strongly correlated with each other (Fig. 2; Spearman rank order correlation, R > 0.80). In general, these metrics of development were inversely correlated

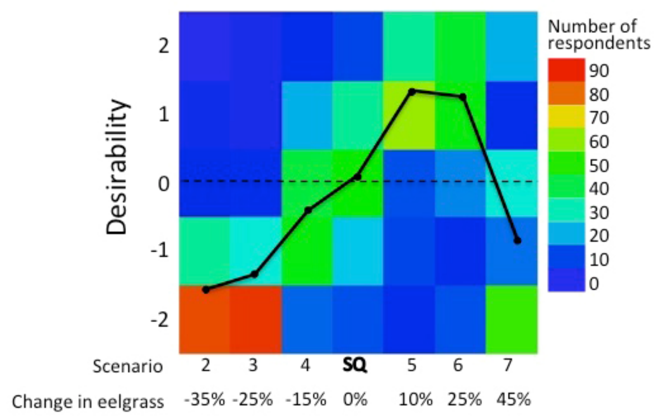
Fig. 3. Participants were exposed to the scenarios in two ways. In addition to visualizations, (Fig. 2), we first used radar plots showing relative trade-offs among 16 different attributes for 6 different scenarios (see Table 1). Values are plotted relative to the status quo (scenario 1), which is depicted as the gray circle in each plot. Biological ecosystem components are shown in blue; costs are in green; yellow, orange, and red depict anthropogenic pressures and some geographic attributes of the region.



with costs (Fig. 2; Spearman rank order correlation, R < –0.65). Increases in eelgrass were generally associated with increasing costs (R = 0.43), decreases in the metrics of development (R > 0.90), and increases in urban density (R > 0.90; Fig. 2). Overall, our scenarios portrayed positive relationships of eelgrass with iconic species of the region including bald eagles, Pacific salmon, and Pacific herring (R> 0.90), while representing an inverse association of gulls with eelgrass (Fig. 2).

When shown the radar plots and stylized images representing the seven scenarios, participants revealed a strong negative reaction to the two most aggressive unconstrained growth scenarios (Fig. 4). Eelgrass in these scenarios declined 35% and 25%, and more than 67% of the participants rated these scenarios as highly undesirable (i.e., –2). Desirability ratings were highest (median = 1.0) for scenarios 5 and 6, in which eelgrass increased by 10% and 25%, respectively. Agreement among participants was generally higher for scenarios that involved eelgrass destruction versus restoration; the interquartile range for scenarios 2, 3, and 4 was 1.0, whereas for the status quo scenario and scenarios 5, 6, and 7 the interquartile range was 2.0. Overall, the preferred state was an increase of between 10% and 25% of eelgrass, whereas the status quo was the minimally acceptable condition.

Fig. 4. A social norm curve showing desirability of seven development scenarios (and associated changes in eelgrass) on a Likert scale from –2 (completely unacceptable) to +2, (optimal state). The line depicts the average desirability of each scenario; the colors show the frequency distribution of responses to each scenario.



CONCLUSIONS

The oceans are beset with wicked problems, i.e., problems with a plurality of legitimate perspectives (Ludwig 2001) but no clear right or wrong answers (Jentoft and Chuenpagdee 2009). Such wickedness provides a clear challenge for the development of conservation targets, because implementing reference points represents a crossroads where policy makers face a potentially staggering array of choices. Trade-offs among different components of the social-ecological system may hinder simultaneously achieving all societal goals; accordingly, science could inform decision making by underscoring the costs and benefits of specific targets and how these are distributed among stakeholders. Such information, in the hands of the appropriate decision makers and discussed in the appropriate forums, allows for the informed participation of citizens in the development of social-ecological targets that reflect their values.

We contend that by soliciting input about targets from those who are affected by conservation actions, and bounding this input by what is ecologically achievable and compatible, it will be possible to move forward with holistic management. The approach illustrated here provides a transparent means to solicit stakeholder participation and input for setting conservation targets. Our case study results show that desirability peaks at a social-ecological state that includes a 10%-25% increase in eelgrass, despite the social and economic costs associated with this level of restoration. Thus, a 10%-25% increase in eelgrass may be a sensible initial conservation target. Importantly, desirability for a social-ecological state in which there are no people or associated development is low. Not surprisingly, then, it appears that although respondents would like to see improvements in the ecological components of the system, they are not willing to completely remove people from the system.

It is a truism that if we do not know where we want to go, we will surely have a hard time getting there. Perhaps equally as axiomatic is the fact that if a broad constituency does not contribute to defining the destination, the road will be very bumpy. We hope that the framework we illustrate here will provide a smoother path forward to the development of scientifically rigorous and socially salient conservation targets.

Responses to this article can be read online at:
http://www.ecologyandsociety.org/issues/responses.php/7866

Acknowledgments:

We thank staff of the Puget Sound Partnership and M. Greenberg for their help throughout this project. We appreciate the efforts of A. Platyrhynchos that kept the project from running afoul. This work was supported by Environmental Protection Agency Grant 13-923277-01, as well as a contribution from the Gordon and Betty Moore Foundation Ocean Tipping Points project.

LITERATURE CITED

Bateman, I. J., B. H. Day, A. P. Jones, and S. Jude. 2009. Reducing gain-loss asymmetry: a virtual reality choice experiment valuing land use change.*Journal of Environmental Economics and Management* 58(1):106-118. http://dx.doi.org/10.1016/j.jeem.2008.05.003

Bell, J. 1984. The effect of presentation form on the use of information in annual reports. *Management Science* 30:169-185. http://dx.doi.org/10.1287/mnsc.30.2.169

Bolte, J., and K. Vache. 2010. *Envisioning Puget Sound alternative futures*. PSNERP Final Report. Puget Sound Nearshore Ecosystem Restoration Project, Olympia, Washington, USA. [online] URL: http://envision.bioe.orst.edu/StudyAreas/PugetSound/PSNERP_Final_Report.pdf

Byers, B. A., R. N. Cunliffe, and A. T. Hudak. 2001. Linking the conservation of culture and nature: a case study of sacred forests in Zimbabwe. *Human Ecology* 29:187-218. http://dx.doi.org/10.1023/A:1011012014240

Carey, M. P., P. S. Levin, H. Townsend, T. J. Minello, G. R. Sutton, T. B. Francis, C. J. Harvey, J. E. Toft, K. K. Arkema, J. L. Burke, C.-K. Kim, A. D. Guerry, M. Plummer, G. Spiridonov, and M. Ruckelshaus. 2013. Characterizing coastal foodwebs with qualitative links to bridge the gap between the theory and the practice of ecosystem-based management. *ICES Journal of Marine Science* 71(3):713-724. http://dx.doi.org/10.1093/icesjms/fst012

Carson, R. T., N. E. Flores, and N. F. Meade. 2001. Contingent valuation: controversies and evidence. *Environmental and Resource Economics* 19:173-210. http://dx.doi.org/10.1023/A:1011128332243

Carwardine, J., C. J. Klein, K. A. Wilson, R. L. Pressey, and H. P. Possingham. 2009. Hitting the target and missing the point: target-based conservation planning in context. *Conservation Letters* 2:4-11. http://dx.doi.org/10.1111/j.1755-263X.2008.00042.x

Chan, K. M. A., A. D. Guerry, P. Balvanera, S. Klain, T. Satterfield, X. Basurto, A. Bostrom, R. Chuenpagdee, R. Gould, B. S. Halpern, N. Hannahs, J. Levine, B. Norton, M. Ruckelshaus, R. Russell, J. Tam, and U. Woodside. 2012*b*. Where are *cultural* and *social* in ecosystem services? A framework for constructive engagement. *BioScience* 62:744-756. http://dx.doi.org/10.1525/bio.2012.62.8.7

Chan, K. M. A., T. Satterfield, and J. Goldstein. 2012*a*. Rethinking ecosystem services to better address and navigate cultural values. *Ecological Economics* 74:8-18. http://dx.doi.org/10.1016/j.ecolecon.2011.11.011

Christensen, V., and C. J. Walters. 2004. Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling* 172:109-139. http://dx.doi.org/10.1016/j.ecolmodel.2003.09.003

Christensen, V., C. J. Walters, and D. Pauly. 2005. *Ecopath with Ecosim: a user's guide*. Fisheries Centre, University of British Columbia, Vancouver, British Columbia, Canada.

Di Minin, E., and A. Moilanen. 2012. Empirical evidence for reduced protection levels across biodiversity features from target-based conservation planning. *Biological Conservation* 153:187-191. http://dx.doi.org/10.1016/j.biocon.2012.04.015

Donatuto, J., E. E. Grossman, J. Konovsky, S. Grossman, and L. W. Campbell. 2014. Indigenous community health and climate change: integrating biophysical and social science indicators. *Coastal Management* 42:355-373. http://dx.doi.org/10.1080/08920753.2014.923140

Essington, T., T. Klinger, T. Conway-Cranos, J. Buchanan, A. James, J. Kershner, I. Logan, and J. West. 2011. The biophysical condition of Puget Sound: biology. *Puget Sound Science Review*. [online] URL: http://www.eopugetsound.org/science-review/biophysical-condition-puget-sound-biology

Fiore, S. M., G. W. Harrison, C. E. Hughes, and E. E. Rutström. 2009. Virtual experiments and environmental policy. *Journal of Environmental Economics and Management* 57:65-86. http://dx.doi.org/10.1016/j.jeem.2008.08.002

Fulton, E. A., J. S. Link, I. C. Kaplan, M. Savina-Rolland, P. Johnson, C. Ainsworth, P. Horne, R. Gorton, R. J. Gamble, A. D. M. Smith, and D. C. Smith. 2011. Lessons in modelling and management of marine ecosystems: the Atlantis experience. *Fish and Fisheries* 12:171-188. http://dx.doi.org/10.1111/j.1467-2979.2011.00412.x

Geisler, C. 2010. Must biodiversity hot-spots be social not-spots? Win-win ecology as sustainable social policy. *Journal of Sustainable Development* 4:119-133.

Gray, S., A. Chan, D. Clark, and R. Jordan. 2012. Modeling the integration of stakeholder knowledge in social-ecological decision-making: benefits and limitations to knowledge diversity. *Ecological Modelling* 229:88-96. http://dx.doi.org/10.1016/j.ecolmodel.2011.09.011

Guerry, A. D., M. H. Ruckelshaus, K. K. Arkema, J. R. Bernhardt, G. Guannel, C.-K. Kim, M. Marsik, M. Papenfus, J. E. Toft, G. Verutes, S. A. Wood, M. Beck, F. Chan, K. M. A. Chan, G. Gelfenbaum, B. D. Gold, B. S. Halpern, W. B. Labiosa, S. E. Lester, P. S. Levin, M. McField, M. L. Pinksy, M. Plummer, S. Polasky, P. Ruggiero, D. A. Sutherland, H. Tallis, A. Day, and J. Spencer. 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. *International Journal of Biodiversity Science, Ecosystem Services & Management* 8:107-121. http://dx.doi.org/10.1080/21513732.2011.647835

Haab, T. C., and K. E. McConnell. 2002. *Valuing environmental and natural resources: the econometrics of non-market valuation*. Edward Elgar, Cheltenham, UK. http://dx.doi.org/10.4337/9781843765431

Harvey, C. J., G. D. Williams, and P. S. Levin. 2012. Food web structure and trophic control in central Puget Sound. *Estuaries and Coasts* 35:821-838. http://dx.doi.org/10.1007/s12237-012-9483-1

Hering, D., A. Borja, J. Carstensen, L. Carvalho, M. Elliott, C. K. Feld, A.-S. Heiskanen, R. K. Johnson, J. Moe, D. Pont, A. L. Solheimh, and W. van de Bundj. 2010. The European Water Framework Directive at the age of 10: a critical review of the achievements with recommendations for the future. *Science of the Total Environment* 408:4007-4019. http://dx.doi.org/10.1016/j.scitotenv.2010.05.031

Hicks, C. C., T. R. McClanahan, J. E. Cinner, and J. M. Hills. 2009. Trade-offs in values assigned to ecological goods and services associated with different coral reef management strategies. *Ecology and Society* 14(1)10. [online] URL: http://www.ecologyandsociety.org/vol14/iss1/art10/

Jackson, J. 1965. Social stratification, social norms, and roles. Pages 301-309 *in* I. D. Steiner and M. Fishbein, editors. *Current studies in social psychology.* Holt, Rinehart & Winston, New York, New York, USA.

Jentoft, S., and R. Chuenpagdee. 2009. Fisheries and coastal governance as a wicked problem. *Marine Policy* 33:553-560. http://dx.doi.org/10.1016/j.marpol.2008.12.002

Kaplan, I. C., P. J. Horne, and P. S. Levin. 2012. Screening California current fishery management scenarios using the Atlantis end-to-end ecosystem model. *Progress in Oceanography* 102:5-18. http://dx.doi.org/10.1016/j.pocean.2012.03.009

Krutilla, J. V. 1967. Conservation reconsidered. *American Economic Review* 57:777-786.

Lackey, R. T. 1998. Seven pillars of ecosystem management. *Landscape and Urban Planning* 40:21-30. http://dx.doi.org/10.1016/S0169-2046(97)00095-9

Large, S. I., G. Fay, K. D. Friedland, and J. S. Link. 2013. Defining trends and thresholds in responses of ecological indicators to fishing and environmental pressures. *ICES Journal of Marine Science* 70:755-767. http://dx.doi.org/10.1093/icesjms/fst067

Levin, P. S. 2014. New conservation for the Anthropocene Ocean. *Conservation Letters* 7:339-340. http://dx.doi.org/10.1111/conl.12108

Levin, P. S., I. Kaplan, R. Grober-Dunsmore, P. M. Chittaro, S. Oyamada, K. Andrews, and M. Mangel. 2009. A framework for assessing the biodiversity and fishery aspects of marine reserves. *Journal of Applied Ecology* 46:735-742. http://dx.doi.org/10.1111/j.1365-2664.2009.01667.x

Ludwig, D. 2001. The era of management is over. *Ecosystems* 4:758-764. http://dx.doi.org/10.1007/s10021-001-0044-x

Moilanen, A., and A. Arponen. 2011. Setting conservation targets under budgetary constraints. *Biological Conservation* 144:650-653. http://dx.doi.org/10.1016/j.biocon.2010.09.006

Mumford, T. F. 2006. *Kelp and eelgrass in Puget Sound*. Puget Sound Nearshore Partnership Report No. 2007-05. Seattle District, U.S. Army Corps of Engineers, Seattle, Washington, USA.

Naidoo, R., A. Balmford, P. J. Ferraro, S. Polasky, T. H. Ricketts, and M. Rouget. 2006. Integrating economic costs into conservation planning. *Trends in Ecology & Evolution* 21:681-687. http://dx.doi.org/10.1016/j.tree.2006.10.003

Nelson, R. H. 2006. Valuing nature. *American Journal of Economics and Sociology* 65:525-557. http://dx.doi.org/10.1111/j.1536-7150.2006.00465.x

Neuendorf, K. A. 2002. *The content analysis guidebook*. First edition. Sage, Thousand Oaks, California, USA.

Ogden, J. C., S. M. Davis, K. J. Jacobs, T. Barnes, and H. E. Fling. 2005. The use of conceptual ecological models to guide ecosystem restoration in South Florida. *Wetlands* 25:795-809. http://dx.doi.org/10.1672/0277-5212(2005)025[0795:TUOCEM]2.0.CO;2

Plagányi, É. 2007. *Models for an ecosystem approach to fisheries*. FAO Fisheries Technical Paper No. 477. Food and Agriculture Organization of the United Nations, Rome, Italy.

Plummer, M. L. 2009. Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment* 7:38-45. http://dx.doi.org/10.1890/080091

Plummer, M. L., C. J. Harvey, L. E. Anderson, A. D. Guerry, and M. H. Ruckelshaus. 2013. The role of eelgrass in marine community interactions and ecosystem services: results from ecosystem-scale food web models. *Ecosystems* 16:237-251. http://dx.doi.org/10.1007/s10021-012-9609-0

Poe, M. R., K. C. Norman, and P. S. Levin. 2014. Cultural dimensions of socioecological systems: key connections and guiding principles for conservation in coastal environments. *Conservation Letters* 7:166-175 http://dx.doi.org/10.1111/conl.12068

Pressey, R. L., R. M. Cowling, and M. Rouget. 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Floristic Region, South Africa. *Biological Conservation* 112:99-127. http://dx.doi.org/10.1016/S0006-3207(02)00424-X

Rehr, A. P., G. D. Williams, and P. S. Levin. 2014*b*. A test of the use of computer generated visualizations in support of ecosystem-based management. *Marine Policy* 46:14-18. http://dx.doi.org/10.1016/j.marpol.2013.12.012

Rehr, A. P., G. D. Williams, N. Tolimieri, and P. S. Levin. 2014*a*. Impacts of terrestrial and shoreline stressors on eelgrass in Puget Sound: an expert elicitation. *Coastal Management* 42:246-262. http://dx.doi.org/10.1080/08920753.2014.904195

Rockeach, M. 2008. *Understanding human values*. Free Press, New York, New York, USA.

Safford, T. G., K. C. Norman, M. Henly, K. E. Mills, and P. S. Levin. 2014. Environmental awareness and public support for protecting and restoring Puget Sound. *Environmental Management* 53:757-768. http://dx.doi.org/10.1007/s00267-014-0236-8

Samhouri, J. F., S. E. Lester, E. R. Selig, B. S. Halpern, M. J. Fogarty, C. Longo, and K. L. McLeod. 2012. Sea sick? Setting targets to assess ocean health and ecosystem services. *Ecosphere* 3:art41. http://dx.doi.org/10.1890/es11-00366.1

Samhouri, J. F., P. S. Levin, and C. H. Ainsworth. 2010. Identifying thresholds for ecosystem-based management. *PLoS One* 5(1):e8907. http://dx.doi.org/10.1371/journal.pone.0008907

Sanderson, E. W. 2006. How many animals do we want to save? The many ways of setting population target levels for conservation. *BioScience* 56:911-922. http://dx.doi.org/10.1641/0006-3568(2006)56[911:HMADWW]2.0.CO;2

Svarstad, H., L. K. Petersen, D. Rothman, H. Siepel, and F. Wätzold. 2008. Discursive biases of the environmental research framework DPSIR. *Land Use Policy* 25:116-125. http://dx.doi.org/10.1016/j.landusepol.2007.03.005

Tallis, H., S. E. Lester, M. Ruckelshaus, M. Plummer, K. McLeod, A. Guerry, S. Andelman, M. R. Caldwell, M. Conte, S. Copps, D. Fox, R. Fujita, S. D. Gaines, G. Gelfenbaum, B. Gold, P. Kareiva, C.-K. Kim, K. Lee, M. Papenfus, S. Redman, B. Silliman, L. Wainger, and C. White. 2012. New metrics for managing and sustaining the ocean's bounty. *Marine Policy* 36:303-306. http://dx.doi.org/10.1016/j.marpol.2011.03.013

Tear, T. H., P. Kareiva, P. L. Angermeier, P. Comer, B. Czech, R. Kautz, L. Landon, D. Mehlman, K. Murphy, M. Ruckelshaus, J. M. Scott, and G. Wilhere. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. *BioScience* 55:835-849. http://dx.doi.org/10.1641/0006-3568(2005)055[0835:HMIETR]2.0.CO;2

Thrush, C.-P. 2009. Native Seattle: histories from the crossing-over place. University of Washington Press, Seattle, Washington, USA.

Uusitalo, L. 2007. Advantages and challenges of Bayesian networks in environmental modelling. *Ecological Modelling* 203:312-318. http://dx.doi.org/10.1016/j.ecolmodel.2006.11.033

Vaske, J. J., M. P. Donnelly, and B. Shelby. 1993. Establishing management standards: selected examples of the normative approach. *Environmental Management* 17:629-643. http://dx.doi.org/10.1007/BF02393725

Watson, M. 2003. Performing place in nature reserves. *Sociological Review* 51:145-160. http://dx.doi.org/10.1111/j.1467-954X.2004.00456.x