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ECOSYSTEMS

A Public Sentiment Index for Ecosystem Management

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Abstract

Although ecosystem-based management can lead to sustainable resource use, its successful implementation depends on stakeholders' acceptance. A framework to integrate scientific knowledge about the ecosystems with stakeholders' preferences is therefore needed. We propose here a 'Public Sentiment Index,' or PSI, as an integration framework that combines an ecosystem model (Ecopath with Ecosim; EwE) with a public choice model (the damage schedule). Using Chesapeake Bay as a case study, we demonstrate the development of the PSI, based on judgments of Bay stakeholders, including 'watermen' (commercial fishers), seafood wholesalers and retailers, recreational fishers, representatives from non-governmental organizations, scientists and managers on a range of Bay ecosystems. The high PSI for Chesapeake Bay suggests a consensus amongst Bay stakeholders who, understanding the need for restoring the Bay ecosystem, may accept difficult policy choices and support their implementation.

Key words: ecosystem management; Chesapeake Bay; ecosystem modeling; paired comparison survey; stakeholders' preferences; public sentiment index.

Introduction

The recent interest in ecosystem-based management of fisheries results from increasing knowledge on the dynamics of trophic interactions between species (Pauly and others 1998), the impacts of fishing on habitats (Dayton and others 1995), and the overall impacts on ecosystems (Chuenpagdee and others 2003). Ecosystem-based management is one of the approaches that may lead to reducing the likelihood of fisheries collapses, and contributing to ecosystem rebuilding and conservation (NRC 1999; Pauly and others 2002; WWF 2002). The applicability of ecosystem-based management has been questioned, however, because of the difficulty in decision making regarding entire ecosystems and in implementation of new management schemes. In the US, for example, fisheries ecosystem plans (FEP) are being slowly formulated and implemented based on the recommendations by the Ecosystem Principles Advisory Panel (EPAP 1999). One case is the Chesapeake Bay (Figure 1), the largest estuary in the US, with a history of successive fisheries depletions. It took about 3 years to complete the drafting process of the FEP, and the Chesapeake Bay FEP is currently being implemented for five species, that is, blue crab, oyster, menhaden, striped bass and shad (see a pre-publication copy at the bottom of the link http://noaa.chesapeakebay.net/Fish/default.htm).

Single-species and ecosystem-based management operate on different scales, and aim at different things. The former aims mainly at estimating total allowable catches (TAC), which require detailed information and high precision, but a relatively short time scale. Ecosystem-based management, on the other hand, provides the framework within

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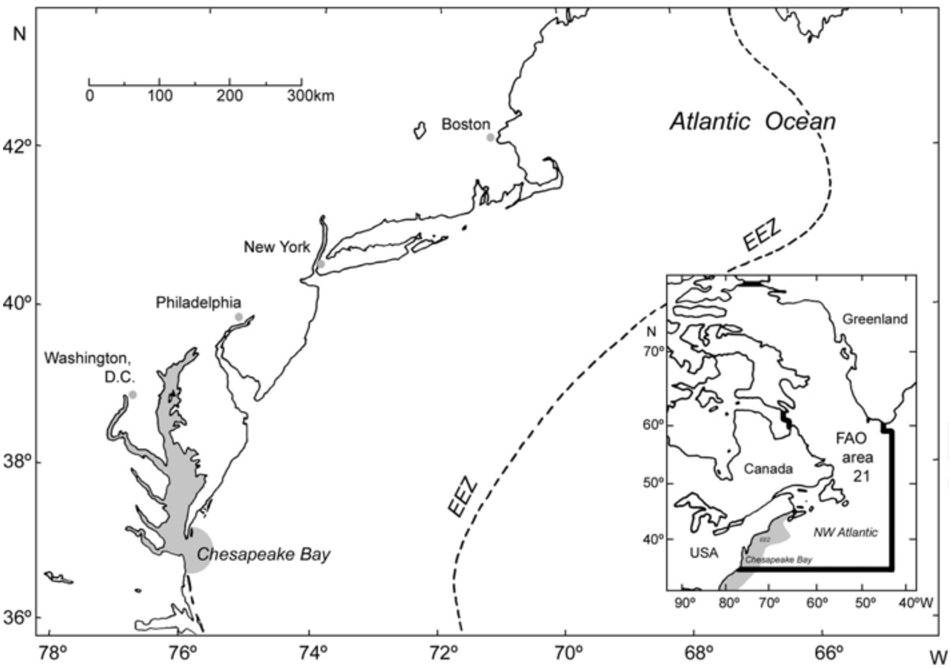

Figure 1. Map of Chesapeake Bay and adjacent shelf waters, nested in the US east coast, from North Carolina to the Gulf of Maine in the North, and itself part of FAO Area 21, including the US coast north of North Carolina, the east coast of Canada and the West coast of Greenland.

which TAC are evaluated, and hence require basic information, of lower precision, but a longer time scale. This is illustrated by the study of Pauly and others (1998), that documented, based on catch and trophic level (TL) estimates, a global trend toward lower TL in fisheries catches, suggesting a lack of sustainability. This study and others for various locations, for example, Canadian waters (Pauly and others 2001), Greek waters (Stergiou and Koulouris 2000), Celtic Sea (Pinnegar and others 2002), Icelandic waters (Valtýsson and Pauly 2003), and Gulf of Thailand (Pauly and Chuenpagdee 2003), demonstrate that the status of any fisheries ecosystem can be inferred straightforwardly, provided that time series of landing data exist for the main species groups, and broad-based TL estimates are available. These examples, drawn upon readily available sources like FishBase (www.fishbase.org), containing biological and ecological information of all exploited marine fish species, illustrate that trends of mean TL of fisheries catches can be analyzed through a metric that quantifies the ecosystem impact of fishing (Pauly and others 1998).

Determining the changes in an ecosystem is only the first step. As suggested by Pitcher (2001), halting and reversing the perverse trend of fisheries require drastic reduction of fishing effort on key species, thus entailing changes from the way fisheries have been traditionally managed. More importantly, implementing an ecosystem-based management plan is not likely to be successful without consultation with stakeholders. As shown in numerous studies on co-management and community-based management (for example, Jentoft and McCay 1995; Sen and Nielsen 1996; Luttinger 1997; Ellsworth and others 1997), meaningful stakeholder participation is a prerequisite for effective policy implementation. It is under this premise that the following study on the Chesapeake Bay stakeholders' preferences was conducted. Here, we propose the use of a 'public sentiment index' or PSI to measure the degree of acceptability of various fishery policy options, and thus the likelihood of implementation success, based on knowledge, judgments, and preferences of stakeholders about an ecosystem.

We first describe, in the Materials and Methods section, how the trend of mean TL in fisheries catches is analyzed and used to suggest changes in the Bay ecosystem from 1950 to 2000. This is followed by the description of how the ecosystem model, 'Ecopath with Ecosim' (EwE; see www.ecopath.org), is employed to generate a set of hypothetical, but realistic ecosystem scenarios, each based on a different abundance level in four major species groups, that is, striped bass, menhaden, blue crab and oysters. Next, we present the 'damage schedule approach' (Chuenpagdee and others 2001a, b), as a tool to elicit stakeholders' preferences about the Bay ecosystem. The method involves use of the paired comparison survey to present these ecosystem scenarios as binary choices to the Bay stakeholders. We explain how the survey results are used to construct the damage schedule and the PSI. The result of the analysis of mean TL trend is presented in the next section, followed by the results of the stakeholders' survey and the PSI for Chesapeake Bay. In the final sections, we discuss the validity of the methods and the interpretation of the results, and conclude with the relevance of the PSI as a tool to help prioritize among a set of fisheries policies for Chesapeake Bay.

Materials and Methods

Measuring Ecosystem Changes in Chesapeake Bay

Chesapeake Bay is home to a variety of large marine fishes, vertebrates and invertebrates, although some are extremely scarce, like oysters (*Crassostrea virginica*), or extinct, like Atlantic gray whales. This elimination of species at the bottom and the top of the Bay's food web, due largely to overfishing and habitat destruction, has induced numerous changes in ecosystem function. Although the fate of large long-lived organisms is widely accepted as irrevocable (owing to 'shifting baselines', Pauly 1995), much controversy surrounds the best mechanisms for rebuilding the oysters and blue crab (*Callinectes sapidus*) stocks in Chesapeake Bay. As one study indicates, the declining spawning stock, larval abundance and mean size of the blue crab are unlikely to rebound without significant reduction in fishing and natural mortality (Lipcius and Stockhausen 2002).

Ecosystem changes in Chesapeake Bay can be measured based on the trend of mean TL in fisheries catches. This was done by first extracting landings data from within Chesapeake Bay and the coastal waters of Maryland and Virginia from 1950 to 2000 from commercial catch and recreational databases maintained by National Marine Fisheries Service (NMFS) (www.st.nmfs.gov/st1/index.html). These landings consist of menhaden (*Brevoortia tyrannus*), other finfish (consisting of 129 named species, plus 21 higher taxa), blue crab, oyster, and other invertebrates (18 named species, plus 8 higher taxa). Next, the average TL was calculated for each species based on diet composition data extracted from FishBase and local sources, notably Hagy (2002), who reviewed a large amount of literature on the diet composition of menhaden, blue crab and other important species in Chesapeake Bay, and used it to define diet compositions included in a network analytic model (see also Ulanowicz 1986).

Chesapeake Bay Ecosystem Model and Scenarios Development

The model of the Bay ecosystem used here was the Ecopath with Ecosim model (EwE; Christensen and Walters 2004; Pauly and others 2000), jointly constructed by a group of locally-based and external scientists and modelers (Christensen and others 2004). The EwE model of Chesapeake Bay consists of 46 groups of fish, vertebrates and invertebrates, and submerged aquatic vegetation (SAV). For the main species groups, including striped bass (*Morone saxatilis*), menhaden, blue crab and oysters, the model takes into consideration different life history stages into account for the different ecosystem roles they perform as juveniles, and resident or migratory adults. Other organisms of less relevance to policy issues were aggregated, for example, as 'reef-associated fish' or 'demersal infauna and epifauna'. All available time series data on relative abundance and related statistics of Bay species from 1950 to 2000 were compiled from sources such as federal and state agencies, and research institutions linked to Chesapeake Bay.

The Bay EwE model used as a basis for this study (Christensen and others 2004) was developed through an 18-month period of consultation and discussion at several EwE Chesapeake Bay workshops, and a rigorous, iterative revision process. Although the model remains under development, it can be used in a heuristic analysis of stakeholder responses to a diverse set of policy and ecosystem options. In general, we found that the model was able to capture the key aspects (both in direction and magnitude) of historical biomass changes, particularly in the four species, that is, striped bass, menhaden, blue crab and oysters, for which detailed information was available. These four species are also of major interest to different stakeholder groups, as indicated in the stakeholders' workshops, and thus were chosen for further analysis.

The year 2000 state of the system, described by EwE, was used to project the biomass of striped bass, menhaden, blue crab and oyster in the year 2050, under 21 different fishing policies, involving increase, decrease or no change in fishing rates of overall species, menhaden or blue crab (Table 1). Further, each fishing policy was overlaid by three policies concerning changes in SAV area (that is, increase, decrease or no change). In both cases, for simplicity, only incremental change of 50 and 100% were used. Although hypothetical, these policies are realistic, that is, they were formulated based on recommendations and roundtable discussion at the second EwE Chesapeake Bay workshop (held at the Virginia Institute of Marine Science, Virginia, USA, in May 2002, and attended by 20 Bay scientists and managers). A total of 21 possible scenarios of biomass estimates for each of

Table 1. Ecosystem Scenarios Generated by the EwE Chesapeake Bay Simulation Model

	Fishing policies		Predicted biomass (in 2050)			
Scenario no.	Fishing rates	SAV	Striped bass	Menhaden	Blue crab	Oyster
1A	**n/c**	**n/c**	**− 25%**	**+ 150%**	**n/c**	**+ 350%**
1B	n/c	+ 100%	+ 25%	+ 200%	n/c	+ 150%
1C	n/c	− 50%	− 75%	+ 150%	n/c	+ 500%
2A	**− 50% (overall)**	**n/c**	**+ 25%**	**+ 200%**	**n/c**	**+ 350%**
2B	− 50% (overall)	+ 100%	+ 50%	+ 150%	n/c	+ 100%
2C	− 50% (overall)	− 50%	− 25%	+ 150%	n/c	+ 350%
3A	**+ 100% (overall)**	**n/c**	**− 75%**	**+ 150%**	**n/c**	**+ 250%**
3B	+ 100% (overall)	+ 100%	− 50%	+ 150%	n/c	+ 50%
3C	+ 100% (overall)	− 50%	− 75%	+ 150%	n/c	+ 350%
4A	**− 50% (menhaden)**	**n/c**	**− 25%**	**+ 200%**	**n/c**	**+ 350%**
4B	− 50% (menhaden)	+ 100%	+ 25%	+ 150%	n/c	+ 100%
4C	− 50% (menhaden)	− 50%	− 50%	+ 150%	n/c	+ 350%
5A	+ 100% (menhaden)	n/c	− 25%	+ 50%	n/c	+ 500%
5B	+ 100% (menhaden)	+ 100%	n/c	+ 100%	n/c	+ 350%
5C	**+ 100% (menhaden)**	**− 50%**	**− 75%**	**+ 100%**	**n/c**	**+ 450%**
6A	− 50% (blue crab)	n/c	− 25%	+ 100%	+ 25%	+ 900%
6B	**− 50% (blue crab)**	**+ 100%**	**+ 25%**	**+ 100%**	**+ 25%**	**+ 900%**
6C	**− 50% (blue crab)**	**− 50%**	**− 75%**	**+ 100%**	**+ 25%**	**+ 900%**
7A	**+ 100% (blue crab)**	**n/c**	**− 25%**	**+ 50%**	**− 75%**	**+ 900%**
7B	**+ 100% (blue crab)**	**+ 100%**	**+ 25%**	**+ 100%**	**− 75%**	**+ 900%**
7C	+ 100% (blue crab)	− 50%	− 75%	+ 50%	− 100%	+ 900%

n/c = No change; −x % ... (y) in fishing rates means a percentage decrease in fishing rates of species group 'y'; '+ x % (y)' in fishing rates means a percentage increase in fishing rates of species group 'y'; percentage changes in SAV areas were indicated by '− or +' sign. Bold scenarios indicated those used in the paired comparison survey.

the four species for the year 2050 were generated (Table 1).

It is notable that all biomass estimates for menhaden and oysters were positive under the specified fishing policies, whereas blue crab biomass was affected only by the direct changes in blue crab fishing rates. The sensitivity analysis of the model showed that drastic policies were required to generate stronger (or negative) changes in these three species groups. These policies were not included in the study because they were considered unrealistic and unlikely to be accepted by stakeholders. Although the percent changes might vary slightly with further fine-tuning of the model, the magnitude of the changes will most probably stay the same as projected (V. Christensen, personal communication, January 2005). As seen in the last six scenarios of Table 1 (scenarios 6A–C and 7A–C), oyster abundance was also insensitive to small changes in the fishing rates of blue crab. According to the model, a drastic reduction of blue crabs (that is, killing them all) would be necessary to cause changes in oyster population. It should be noted, however, that the current model does not directly capture the effects of diseases like MSX and Dermo, which are suggested to be one of the major sources of oyster mortality (Sea Grant 2003).

The Damage Schedule Approach and Stakeholders' Preferences Survey

Following the damage schedule approach (Chuenpagdee and others 2001a, b), the method of paired comparison was used to present the scenarios as binary choices. The method of paired comparison was selected for the study because it was found, in many cases, to elicit highly reliable judgments, especially with regard to natural resources and environmental issues (Opaluch and others 1993; Peterson and Brown 1998). For 21 scenarios (n), the total number of all possible pairs (N) for each respondent is $[n\ (n{-}1)/2]$ or 210 pairs. This number was considered too high because earlier studies show that the multitudes of choice sets induce fatigue and inconsistent responses (David 1988; Chuenpagdee and others 2001a, b). Thus, only 9 out 21 ecosystem scenarios were selected for the paired comparison (bold scenarios in Table 1).

The selection of these scenarios was based on the following criteria. First, they had to cover all possible ranges in fishing policies and ecosystem configurations, that is, negative and positive changes were evenly presented. Further, they had to be sufficiently different in at least one dimension (fishing policies or changes in biomass) to enable binary choice responses. These criteria eliminated potential biases in respondents' judgments and increased the efficiency in the selection, respectively.

The respondents were asked to select, for each pair, the ecosystem configuration that they preferred (Figure 2). Note that these pairs were presented without information on the required policy changes so as to allow respondents to concentrate solely on the ecosystem configurations, and not to base their choices on the policies behind the changes. A random order was used to position scenarios as choice A or B, and to present the pairs in the survey booklet, such that each booklet was unique. The former eliminated possible bias that some respondents may have with regard to choices A or B, while the latter removed bias associated with the order of presentation of the scenarios.

Seven groups of stakeholders around Chesapeake Bay were targeted, that is, Virginia and Maryland watermen, recreational fishers, seafood business people, fishery managers, scientists, and non-governmental organizations (NGOs). Here, stakeholders were groups of people with diverse interests on the Bay, starting from those relying economically on the condition of fisheries resources (for example, producers and distributors), people gaining recreational benefits from the Bay, those with scientific knowledge about Chesapeake Bay, to those responsible for managing it. Non-governmental organizations were included in the study to represent the public, which has a general interest in the Bay, but does not depend on it for their livelihood.

A comprehensive list of all stakeholders in each group was compiled using available information from related associations, organizations and institutions. Potential respondents were then randomly selected and contacted by telephone or e-mail about their willingness to complete the survey, until between 35 and 55 people had agreed. The survey booklets (reviewed by the Chesapeake Bay Advisory Committee and approved by an ethical committee for use of human subjects in research and study of the College of William and Mary, Virginia) were then mailed to stakeholders who had agreed to participate, along with a cover letter assuring respondents of confidentiality and anonymity, and a stamped, addressed envelope for returning of the questionnaire. Paired comparison responses indicating ecosystem preferences for each stakeholder group were scored, aggregated and normalized to the scale of 0 to 100 (Dunn-Rankin 1983). A rank order was assigned to these preference scores such that '1' referred to the most preferred scenario and '9' the least preferred, and Kendall's tau rank correlation analysis was performed to determine the ranking agreement.

In addition to the paired-comparison questions, the survey included a section where six out of nine policies were presented to the respondents, who were invited to indicate their acceptability. Three policies removed from this section were the one involving no change (1A), and those involving overall fishing rates (2A and 3A). In the next step, we combined the preference scores from the paired comparison questions with the percentage of policy acceptance, and normalized it to yield a PSI. The final section of the survey contained information about the respondents, such as gender, age, education, occupation, residency, as well as questions about water-related activities conducted in Chesapeake Bay.

Results

Ecosystem Changes in Chesapeake Bay

As shown in Figures 3a and b, data from within Chesapeake Bay and the coastal waters of Maryland and Virginia indicated that landings in the 1950s were comparable between menhaden and other groups, whereas in 2000 menhaden dominated the catches (Figure 3a). Despite the highly fluctuating landings of the main species, a declining trend from 1995 for all groups was observed (Figure 3b).

The mean TL of the catch declined at about 0.05 per decade, that is, from 2.64 in the early 1950s to 2.37 in 2000 (Figure 4). This trend, on the lower end of the estimated range of 0.05–0.1 for ecosystems worldwide, is similar to the trends of TL for the two ecosystems within which the Bay is embedded, that is, the US east coast and mid-Atlantic and FAO Area 21 (Figure 1). Embedding this trend in a broader, regional context is compatible with the hypothesis that, although the recent changes occurring in Chesapeake Bay may be attributed to pollution and disease, historical removal by fishing of a vast number of components from that ecosystem's food web contributed greatly to the current state of the Bay (Jackson and others 2001). Figure 4 shows that in the 1950s–1980s, Chesapeake Bay and the US East coast had much lower TL values than FAO Area 21. In general, the declining mean TL reflects how species susceptible

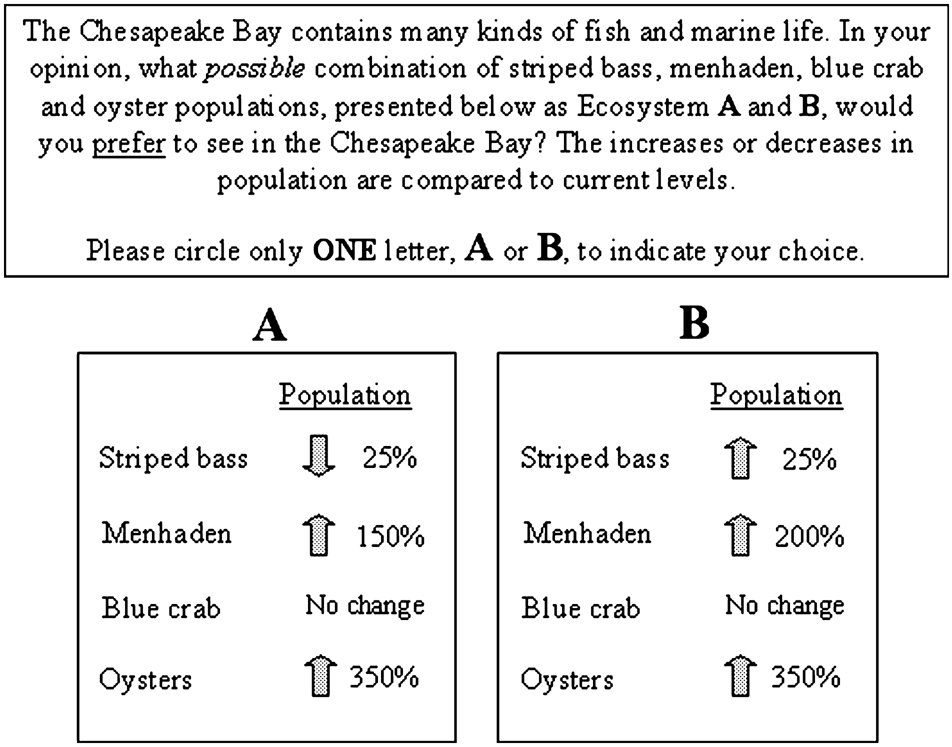

Figure 2. An example of a paired comparison of ecosystem scenarios presented in the survey of Chesapeake Bay stakeholders' preferences.

to overfishing (that is, long-lived, high-TL fishes) are being replaced by less susceptible species (usually forage fish with low TL, and invertebrate), as is the case of FAO Area 21, which was previously dominated by catches of cod and other high-TL fishes (Pauly and others 2001). The downward trend in TL in Chesapeake Bay, moreover, occurred in spite of the decline in the lower TL species, particularly oyster and crab. Overall, the TL analysis clearly shows the ecosystem changes that took place in Chesapeake Bay, and supports the need for restoration and conservation effort.

PSI for Chesapeake Bay

Of a total of 370 surveys sent, 159 were returned, resulting in a 43% response rate. The analysis was based on 125 completed surveys, ranging between 11 and 25 respondents in each stakeholder group (Table 2). Kendall's tau rank correlation analysis showed that the rankings of ecosystem preference by the seven stakeholder groups were not significantly different (Table 2). Thus, all responses were aggregated to yield one set of preference scores and ranking (Table 3). Overall, the most preferred ecosystem scenario (6B) consisted, unsurprisingly, of increases of biomass in all four species groups, whereas the least preferred (7A) involved a 75% decrease in blue crab and a 25% decrease in striped bass biomass. In general, the preferred ecosystem scenarios for blue crab were those with an increase or no change in biomass. The preference for striped bass varied depending on the changes in biomass of the other three groups. Thus, a reduction in biomass of striped bass was accepted if accompanied by an increase in blue crab (scenario 6C). On the other hand, a decrease of blue crab was not accepted, even when accompanied by an increase in striped bass (scenario 7B).

As noted earlier, the paired comparison survey elicited stakeholders' preferences for various configurations of the Chesapeake Bay ecosystem without explicit consideration of the policies leading to those configurations. The stated preferences were thus free of the individual bias associated with the perceived personal impact of different policies. Whatever these personal biases may be, public acceptance of a fishing policy is crucial for the success of its implementation. The next section of the survey thus referred to the acceptability of various policies. The results show that about 60% of all respondents accepted policies generating scenario 6B, whereas almost 50% accepted those for scenario 4A, which implied a reduction of menhaden fishing. On the other hand, less than

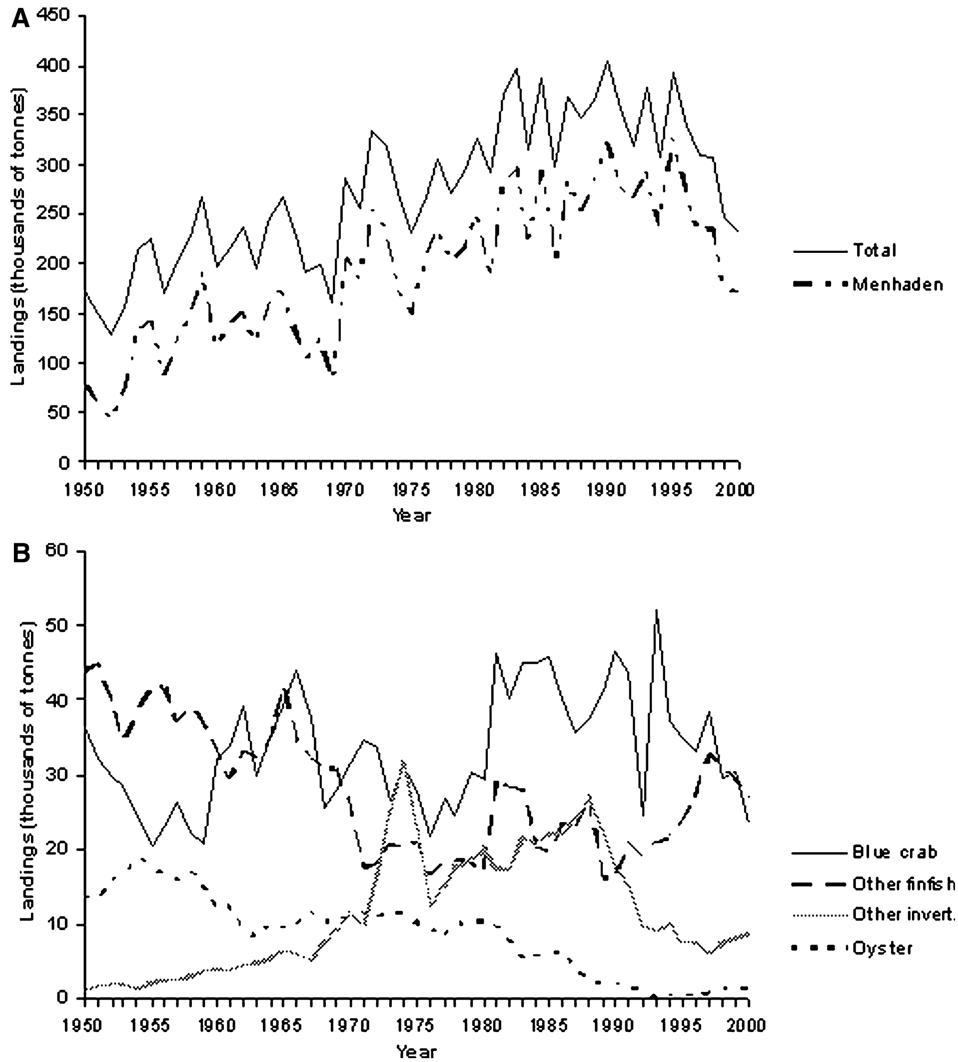

Figure 3. Landings from Chesapeake Bay and adjacent shelf waters, 1950–2000, by major trophic groups: **A** menhaden; and **B** blue crab, oyster, other finfish and other invertebrates.

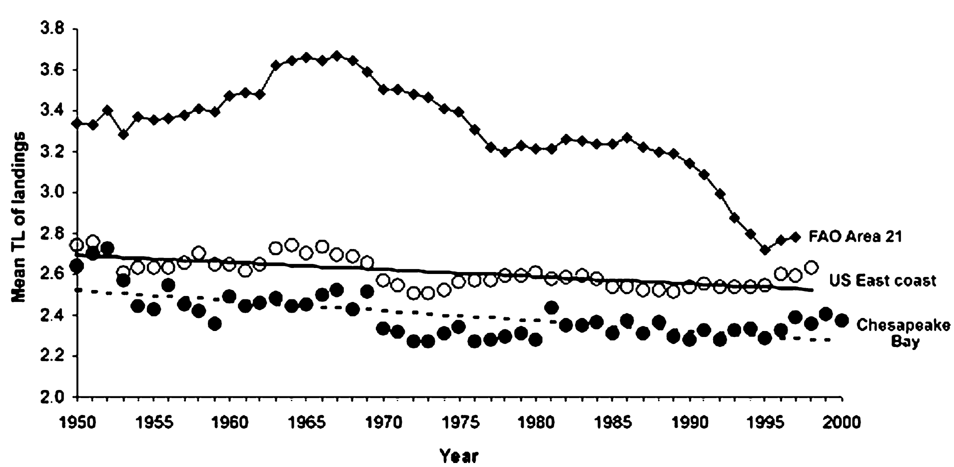

Figure 4. Trends in mean trophic level (*TL*) of the landing in three nested areas of the North Western Atlantic.

Table 2. Preference Scores and Ranking of the Nine Ecosystem Scenarios
(a) Preference scores and (ranking)

Group	Number of respondents	Ecosystem scenarios 1A	2A	3A	4A	5C	6B	6C	7A	7B
VA watermen	13	53 (5)	61 (3)	23 (7)	56 (4)	34 (6)	84 (1)	62 (2)	10 (9)	17 (8)
MD watermen	13	50 (4)	65 (2)	26 (7)	50 (4)	33 (6)	85 (1)	63 (3)	9 (9)	17 (8)
Recreational fishers	23	55 (5)	62 (2)	20 (8)	60 (3)	23 (6)	86 (1)	56 (4)	19 (9)	21 (7)
Seafood business	11	49 (5)	60 (2)	19 (9)	52 (4)	36 (6)	81 (1)	57 (3)	20 (8)	26 (7)
Managers	23	50 (5)	63 (2)	18 (9)	56 (3)	34 (6)	82 (1)	52 (4)	17 (8)	27 (7)
Scientists	25	56 (4)	66 (2)	19 (7)	61 (3)	31 (6)	85 (1)	53 (5)	11 (9)	18 (8)
NGOs	17	46 (4)	71 (2)	16 (9)	54 (3)	25 (7)	88 (1)	43 (5)	22 (8)	35 (6)
Total	125	52 (5)	64 (2)	20 (8)	56 (3)	31 (6)	85 (1)	55 (4)	15 (9)	23 (7)

(b) Kendall's tau rank correlation coefficients showing strong agreement between seven stakeholder groups

	VA watermen	MD watermen	Rec. fishers	Seafood business	Managers	Scientists	NGOs
VA watermen	–						
MD watermen	0.930	–					
Recreational fishers	0.833	0.873	–				
Seafood business	0.833	0.873	0.889	–			
Managers	0.778	0.817	0.944	0.944	–		
Scientists	0.833	0.873	0.889	0.778	0.833	–	
NGOs	0.667*	0.704	0.833	0.833	0.889	0.833	–

VA = Virgrinia, MD = Maryland.
**Correlation is significant at the 0.05 level (two-tailed); all others are significant at the 0.01 level.*
Data were obtained from the survey of 125 Chesapeake Bay stakeholders from seven different groups.

Table 3. A Public Sentiment Index (PSI) for Chesapeake Bay Ecosystem Management

Scenario no.	Fishing policies: Fishing rates	SAV	Total scores	Total ranking	Policy acceptance	PSI
6B	**− 50% (blue crab)**	**+ 100%**	85	1	60.5	72.6
2A	− 50% (overall)	n/c	64	2		
4A	**− 50% (menhaden)**	**n/c**	56	3	48.8	52.5
6C	**− 50% (blue crab)**	**− 50%**	55	4	5.7	30.3
1A	n/c	n/c	52	5		
5C	**+ 100% (menhaden)**	**− 50%**	31	6	2.4	16.5
7B	**+ 100% (blue crab)**	**+ 100%**	23	7	6.6	14.8
3A	+ 100% (overall)	n/c	20	8		
7A	**+ 100% (blue crab)**	**n/c**	15	9	2.5	8.9

Bold denotes policies included in the public acceptance questions.

10% of the respondents indicated acceptance of other policies. The preference scores were combined with the percentage of policy acceptance, and normalized to yield a PSI (see Table 3).

Overall, the PSI, which ranges from 0 to 100, measures the willingness of the stakeholders to follow up on the changes in management and policy that they and others consider necessary. The PSI integrates considerations for ecosystem changes with public preferences for both the changes and policies causing the changes. A high PSI, resulting mostly from a high preference score and high level of acceptance, suggests that there is potential for successful implementation of a particular policy. On the other hand, a low preference score and low level of acceptance yield a low PSI, indicating a low probability of success in implementing a given policy.

In the case of Chesapeake Bay, the policies that can be considered potentially feasible to implement, according to the PSI, are reduction of blue crab fishing with doubling of SAV areas, and reduction of menhaden fishing. Doubling overall fishing rates, even when accompanied by an increase in SAV, yielded a low PSI, suggesting the limited prospect for successful implementation of such policies. As PSI depends on the relative preference scale, it can be easily adjusted to accommodate other changes in the ecosystem. For example, although the survey questionnaire did not include overall reduction of fishing rates as a policy, its acceptance score can be interpolated between 'adjacent' policies with known preference scores.

The ability of PSI to adequately capture the existence of public consensus on policies depends on the public's understanding of ecosystem function, awareness about human impacts, and a general appreciation of the social and economic consequences of policy changes. In our study, about 40% of the respondents lived along the Bay's coastline, 65% fished recreationally, and about 45% were members of environmental organizations. The level of education of the respondents was high, with 23% having a PhD, 14% masters and 18% bachelor degrees, whereas almost all watermen graduated from high school. About 72% of the respondents considered that their daily activities may impact the Bay and 70% indicated that their level of knowledge about the Bay was 'above average'. This establishes that the PSI, in this study at least, is based on responses from informed stakeholders.

The other factor relevant to the PSI is the ability of the ecosystem model, in this case EwE, to predict changes in ecosystem configuration that are perceived as realistic. The levels of abundance indicated in the scenarios must represent the magnitude and direction of changes that are likely to occur, given current information about diet, TL, fishing effort and catches. However, as ecosystem preferences are displayed on an interval scale, it is possible to extrapolate and interpolate other changes not specified in the scenarios.

Discussion and Conclusion

One of the main principles for ecosystem-based fisheries management is to develop a shared vision amongst stakeholders for the use and management of fisheries while considering the value of the ecosystem (EPAP 1999). This generally implies changes in how fisheries are regulated and how policies are formulated, which are often difficult to implement particularly in the system where conflicts between stakeholders exist. As seen in many fisheries around the world, conflicts between stakeholders are caused by increased exploitation and, in some cases, over-exploitation, leading to issues such as access to resources, space, gear conflicts and markets, as illustrated by Nielson and others (2004) for South and Southeast Asia, and by Jentoft (1993) for Norway. Prohibition of access to resources through use of marine protected areas (MPA) is an example of a policy that is contentious with some stakeholders. As stated by Christie and others (2003), controversy and conflicts may arise in the setting of MPA, when biological and social goals are contradictory or unequally appealing to different stakeholder groups. Thus, policy makers, shunning 'unpopular' decisions, may be reluctant to introduce such policies, and often use lack of complete knowledge of the dynamics of the ecosystem as pretext for inaction. While efforts to enhance our understanding of the ecosystems should continue, interim management options should be explored to halt further ecosystem degradation.

Chesapeake Bay is a challenging case for ecosystem management. It has been studied in great detail by generations of scientists, supported by numerous government and private initiatives. In 1983, 35 State and Federal agencies signed the 'Chesapeake Bay Agreement'; 28 other Federal agencies and 12 academic institutions are Chesapeake Bay Program partners, and as many as 500 nonprofit organizations are actively working to restore and conserve natural resources and create sustainable communities in the Chesapeake Bay basin. The large number of organizations and the thousands of individuals in the NGOs who care about Chesapeake Bay indicate the importance of maintaining and restoring the Bay's vital elements, including fisheries and ecosystems that support them.

Nevertheless, the multiplicity of stakeholder groups makes ecosystem management difficult. Chesapeake Bay is home to watermen from Virginia, Maryland and nearby states whose livelihood depends heavily on fishing. Fishing policies that restrict fishing activities will, at least in the short-run, affect the watermen's ability to maintain their catches. Many Bay residents and visitors engage in high levels of recreational fishing, as well as crabbing and clamming. Concentration of residential and urban areas, livestock and poultry farms, forest, and mining industries, and other businesses around the Bay pollute and contaminate its waters. Undoubtedly, fishing and other human activities have had a strong impact on the Bay ecosystems, over a long period of time, and thus mitigating efforts, for example, restoring SAV and oyster population, will also take a long time to yield satisfactory results.

This study of Chesapeake Bay provides an example of what can be done to advance ecosystem management, given existing constraints. First, a diagnosis was performed, here using changes in the mean TL of the catch, to establish that the ecosystem is changing and hence to support ecosystem-based management. Next, the EwE model was used to generate a series of future ecosystem scenarios. The time series data of landings required for the first step are often available from government agencies responsible for fisheries management, whereas the diet composition data for the second step can be obtained from existing sources like FishBase. The model can be improved by better input data, especially with regard to zooplankton and phytoplankton. In addition to the effect of diseases, the model presently does not realistically consider habitat structure, such as the non-trophic effects of SAV and oyster reefs.

Using the results from the EwE model, a stakeholders' preference survey was conducted which suggested desirable ecosystem configurations. The survey was modeled after the damage schedule approach based on the paired comparison responses from different stakeholder groups. The use of the paired comparison method in the damage schedule approach makes it comparable to other methods using multi-attribute analysis such as conjoint analysis and contingent ranking, both of which have been applied to environmental resource settings (for example, Farber and Griner 2000 on watershed quality improvement, and Caplan and others 2002 on curbside waste disposal). Although all three methods rely on binary choices, the damage schedule is considered a 'non-monetary' approach, meaning that it does not imply use of monetary unit as a metric, neither as part of the random utility setting as in conjoint analysis, nor in the willingness-to-pay setting as in contingent ranking. The results of the damage schedule can be considered in the context of cost-benefit analysis only as far as to suggest that the stakeholders' preferences for ecosystem states is an expression of benefits perceived, and the acceptability of policies represents the costs of achieving these states.

The PSI might be useful in a broad framework of public decision-making and policy formulation and design for environmental resources management. In a discussion about good governance, Kooiman (2003) suggested that interactions between stakeholders and the governing institutions are required at all stages in the planning and management. The PSI is a tool that can facilitate such interactions by offering stakeholders a level playing field. For example, all stakeholders, whether experts or lay, are asked the same set of questions and their inputs are equally weighted. Only basic information about the issues of concern is required and it is provided to all respondents regardless of their education and experience. The PSI may be used to provide inputs to policy formulation, as illustrated in this Chesapeake Bay case study, or as inputs for designing an effective implementation plan. The potential use in the latter is based on its ability to address issues causing either real or perceived conflicts. Our study shows that consensus can be found on issues that are typically considered controversial. Such consensus is neither obvious nor guaranteed. Disagreement can be expected and informative, as it indicates issues that should be emphasized and discussed during the policy formulation process.

Overall, the PSI provides a starting point to engage stakeholders in public decision-making, policy formulation and implementation. It offers managers and policy makers an inexpensive tool to prioritize policy options for additional attention. The approach is flexible and is applicable in situations where a combination of scientific and local knowledge and stakeholders' judgments and preferences is required, as is generally the case with management of environmental and natural resources.

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REFERENCES

Caplan AJ, Grijalva TC, Jakus PM. 2002. Waste not or want not? A contingent ranking analysis of curbside waste disposal options. Ecol Econ 43:185–97.

Christensen V, Walters CJ. 2004. Ecopath with ecosim: methods, capabilities and limitations. Ecol Model 172(2–4):109–39.

Christensen V, Beattie A, Buchanan C, Martell SJD, Latour RJ, Preikshot D, Townsend H, Walters CJ. 2004. A dynamic mass-balance model of the Chesapeake Bay ecosystem: methodology, parameterization and model exploration. Draft report for submission to NOAA Tech. Report Series.

Christie P, McCay BJ, Miller ML, Lowe C, White AT, Stoffle R, Fluharty DL, Talaue McManus L, Chuenpagdee R, Pomeroy C, Suman DO, Blount BG, Huppert D, Villahermosa Eisma RL, Oracion E, Lowry K, Pollnac RB. 2003. Toward developing a complete understanding: a social science research agenda for marine protected areas. Fisheries 28(12):22–6.

Chuenpagdee R, Knetsch JL, Brown TC. 2001a. Environmental damage schedules: community judgments of importance and assessment of losses. Land Econ 77(1):1–11.

Chuenpagdee R, Knetsch JL, Brown TC. 2001b. Coastal management using public judgments, importance scales, and predetermined schedule. Coast Manage 29(4):253–270.

Chuenpagdee R, Morgan LE, Maxwell SM, Norse EA, Pauly D. 2003. Shifting gears: assessing collateral impacts of fishing methods in the U.S. waters. Front Ecol Environ 10(1):517–24.

David HA. 1988. The method of paired comparisons. London: Charles Griffin.

Dayton PK, Thrush SF, Agardy MT, Hofman RJ. 1995. Environmental effects of marine fishing. Aquat Conserv: Mar Freshw Ecosyst 5:205–32.

Dunn-Rankin P. 1983. Scaling methods. New Jersey: Lawrence Erlbaum.

Ellsworth JP, Hildebrand LP, Glover EA. 1997. Canada's Atlantic coastal action program: a community-based approach to collective governance. Ocean Coast Manage 36(1–3):121–42.

EPAP 1999. Ecosystem-based fishery management: report to congress. Ecosystem principles advisory panel. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Services. April 1999. 54 p.

Farber S, Griner B. 2000. Valuing watershed quality improvements using conjoint analysis. Ecol Econ 34:63–76.

Hagy JDI. 2002. Eutrophication, hypoxia and trophic transfer efficiency in Chesapeake Bay. PhD Thesis. Baltimore, USA: University of Maryland.

Jackson JB, Kirby MX, Berger WH, Bjorndal KA, Rotsford LW, Bourque BJ, Cooke R, Estes JA, Hughes TP, Kidwell S, Lange CB, Lenihan HS, Pandolfi JM, Peterson CH, Steneck RS, Tegner MJ, Warner RR. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293:629–38.

Jentoft S. 1993. Dangling lines—The fisheries crisis and the future of coastal communities: the Norwegian experience. Social and economic studies no. 50. Institute of Social and Economic Research, Memorial University of Newfoundland. 161 p.

Jentoft S, McCay B. 1995. User participation in fisheries management: lessons drawn from international experiences. Mar Policy 19(3):227–46.

Kooiman J. 2003. Governing and governance. London: Sage.

Lipcius RN, Stockhausen WT. 2002. Concurrent decline of the spawning stock, recruitment, larval abundance, and size of the blue crab *Callinectes sapidus* in Chesapeake Bay. Mar Ecol Prog Ser 226:45–61.

Luttinger N. 1997. Community-based coral reef conservation in the Bay Islands of Honduras. Ocean Coast Manage 36(1–3):11–22.

Nielsen JR, Degnbol P, Viswanathan KK, Ahmed M, Hara M, Abdullah NMR. 2004. Fisheries co-management—an institutional innovation? Lessons from South East Asia and Southern Africa. Mar Policy 28:151–60.

NRC 1999. Sustaining marine fisheries. National Research Council. Washington, D.C: National Academy Press.

Opaluch JE, Swallow SK, Weaver T, Wessells CW, Wichelns D. 1993. Evaluating impacts from noxious facilities: including public preferences in current siting mechanisms. J Environ Econ Manage 24:41–59.

Pauly D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends Ecol Evol 10(10):430.

Pauly D, Chuenpagdee R. 2003. Development of fisheries in the Gulf of Thailand large marine ecosystem: analysis of an unplanned experiment. In: Hempel G, Sherman K, Eds. Large marine ecosystems of the world: trends in exploitation, protection, and research. Amsterdam: Elsevier. pp 337–354.

Pauly D, Christensen V, Dalsgaard J, Froese R, Torres FC Jr. 1998. Fishing down marine food webs. Science 279:860–863.

Pauly D, Christensen V, Walters C. 2000. Ecopath, ecosim, and ecospace as tools for evaluating ecosystem impact of fisheries. ICES J Mar Sci 57:697–706.

Pauly D, Palomares ML, Froese R, Sa-a P, Vakily M, Preikshot D, Wallace S. 2001. Fishing down Canadian aquatic food webs. Can J Fish Aquat Sci 58:51–62.

Pauly D, Christensen V, Guénette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D. 2002. Towards sustainability in world fisheries. Nature 418:689–695.

Peterson GL, Brown TC. 1998. Economic valuation by the method of paired comparison, with emphasis on tests of the transitivity axiom. Land Econ 74(2):240–61.

Pinnegar JK, Jennings S, O'Brien CM, Polunin NVC. 2002. Long-term changes in the trophic level of the Celtic Sea fish community and fish market price distribution. J Appl Ecol 39:377–90.

Pitcher TJ. 2001. Fisheries managed to rebuild ecosystems? reconstructing the past to salvage the future. Ecol Appl 11(2):601–17.

Sea Grant. 2003. Restoring oysters to the U.S. Coastal waters. Maryland and Virginia Sea Grant College Programs for the National Sea Grant College Program.

Sen S, Nielsen JR. 1996. Fisheries co-management: a comparative analysis. Mar Policy 20(5):405–18.

Stergiou KI, Koulouris M. 2000. Fishing down the marine food webs in the Hellenic seas. Fishing down the Mediterranean food webs? In: Durand F, Ed. Proceedings of a CIESM workshop held in Kerkyra, Greece, 26–30 July, 2000. CIESM workshop series No. 12. pp 73–78.

Ulanowicz RE. 1986. Growth and development: ecosystem phenomenology. Berlin Heidelberg New York: Springer.

Valtýsson H, Pauly D. 2003. Fishing down the food web: an Icelandic case study. Competitiveness within the global fisheries. In: Gumundsson E, Valtysson H, Eds. Proceedings of a conference held in Akureyri, Iceland, April 6–7, 2000. University of Akureyri, Akureyri, Iceland. pp 12–24.

WWF 2002. Policy proposals and operational guidance for ecosystem-based management of marine capture fisheries. World Wide Fund for Nature Australia. 83 p.