Challenges for Implementing an Ecosystem Approach to Fisheries Management

Author: Cowan, James H.

Source: Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science, 4(1) : 496-510

Published By: American Fisheries Society

URL: https://doi.org/10.1080/19425120.2012.690825

Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 4:496–510, 2012

ISSN: 1942-5120 online
DOI: 10.1080/19425120.2012.690825

ARTICLE

Challenges for Implementing an Ecosystem Approach to Fisheries Management

James H. Cowan Jr.*
Department of Oceanography and Coastal Sciences, Louisiana State University, Energy, Coast, and Environment Building, Baton Rouge, Louisiana 70803, USA

Jake C. Rice
Fisheries and Oceans Canada, Science Sector, 200 Kent Street, Ottawa, Ontario K1A 0E6, Canada

Carl J. Walters
Fisheries Centre, University of British Columbia, 2202 Main Mall, Vancouver V6T 2K9, Canada

Ray Hilborn and Timothy E. Essington
University of Washington, Box 355020, 1122 Northeast Boat Street, Seattle, Washington 98195, USA

John W. Day Jr.
Department of Oceanography and Coastal Sciences, Louisiana State University, Energy, Coast, and Environment Building, Baton Rouge, Louisiana 70803, USA

Kevin M. Boswell
Marine Sciences Program, Florida International University, 3000 Northeast 151st Street, MSB355, North Miami, Florida 33181, USA

Abstract

The ecosystem approach is being promoted as the foundation of solutions to the unsustainability of fisheries. However, because the ecosystem approach is broadly inclusive, the science for its implementation is often considered to be overly complex and difficult. When the science needed for an ecosystem approach to fisheries is perceived this way, science products cannot keep pace with fisheries critics, thus encouraging partisan political interference in fisheries management and proliferation of "faith-based solutions. In this paper we argue that one way to effectively counter politicization of fisheries decision-making is to ensure that new ecosystem-based approaches in fisheries are viewed only as an emergent property of innovation in science and policy. We organize our essay using three major themes to focus the discussion: empirical, jurisdictional, and societal challenges. We undertake at least partial answers to the following questions: (1) has conventional fisheries management really failed?; (2) can short-comings in conventional fisheries management be augmented with new tools, such as allocation of rights?; (3) is the Ecosystem Approach to Fisheries (EAF) equivalent to Ecosystem-Based Management?; and (4) is restoration of degraded ecosystems a necessary component of an EAF?

Subject editor: Donald Noakes, Thompson Rivers University, British Columbia, Canada

*Corresponding author: jhcowan@lsu.edu
Received March 6, 2012; accepted April 26, 2012

Fisheries management and governance globally underwent a major shift in the second half of the 1970s, with the United Nations Convention on the Law of the Sea (UNCLOS; UN 1995). Although the USA is still not party to UNCLOS, approval of the Magnuson–Stevenson Sustainable Fisheries Act in 1976 (and subsequent reauthorizations, including the Sustainable Fisheries Act in 2006) resulted in similar large policy shifts. Prior to UNCLOS, the chosen scale of management was a matter of convenience, feasibility, and compromise, often regional, sometimes even local. Once adopted, UNCLOS made it legally feasible to manage on scales of stock dynamics out to the full continental shelf, and its application was further extended to highly migratory and straddling stocks by the Fish Stocks Agreement (FSA; UN 1995).

In the years following the extended mandate from UNCLOS, science became an important driver and enabler of change in scale and scope of management, initially through improvements in assessment techniques and data collection (e.g., Browman and Stergiou 2005; Link 2005, 2010; NRC 2006; Fraser et al. 2008; Garcia and Charles 2008; Piet et al. 2008; Salas et al. 2008; Bakun et al. 2009). As improved information about stock delineation and movements, magnitude and nature of fisheries landings, trophic dynamics, and the ecosystem impacts of fisheries became available, assessment processes adopted approaches that allowed consideration of multispecies interactions and the impacts of fishing on habitats and food webs, sometimes at a pace faster than fisheries policy and management were prepared to implement (NRC 2006; Commission of the European Communities [CEC] (2008, (2009). Policy caught up to, and possibly leapt ahead of, science in the late 1990s and 2000s, when several subsequent international policy agreements and declarations, including provisions of the Reykjavik Declaration (FAO 2002b), the St. John's Declaration (2005), and United Nations General Assembly Resolutions 61/105 (UN 2006) and 64/72 (UN 2009); FAO (2003, 2008), further extended UNCLOS and the FSA to broader ecosystem contexts.

In addition, there is a growing sense that traditional management has "failed," although a growing body of literature counters that view. Still, the effect has been to think about alternative approaches, including specific policy instruments such as no-take marine reserves and incentive-based methods, but also larger whole-scale changes in the way the fishery system is perceived and managed. Some proposals are for structural changes, such as simply eliminating large-scale fisheries in favor of small-scale ones (UNEP 2012), but the broader ecosystem mandate for fisheries is getting particular attention as the foundation for solutions to the unsustainability of fisheries (Pikitch et al. 2004). For many, however, there is a disconnect between the past, present, and future concerning how we think about fisheries management, both good and bad. But a sustained disagreement about what has happened in the past, and where we are in the present, is irrelevant—the question now is, what is the best way forward?

The ecosystem approach to fishery has been championed as one way forward because it more holistically considers complex linkages across human and natural systems, identifies conflicts between competing ecosystem services, and also directly considers both direct and indirect impacts of fishing activities on marine ecosystems. However, there is no consensus among experts and critics on how to implement this broader ecosystem mandate, whether through the "ecosystem approach to fisheries" (EAF) or "ecosystem-based management" (EBM); advocates of both approaches have offered few significant new insights concerning details about how they are to be implemented.

Both EAF and EBM are broadly inclusive, so it is easy for advocates to argue they will bring into conventional fisheries management considerations that previously have been overlooked or deemphasized. However, it is not a given that simply adding more considerations to fisheries science and management necessarily makes either of them better or solves problems encountered by conventional practices. The science for implementation of EAF and EBM is often considered to be complex and difficult (Murawski 2007; Link 2010; Rice 2011); indeed, when "ecosystem" is interpreted holistically, the science undoubtedly is complex. When the science needed for EAF or EBM is perceived as including all ecosystem dynamics, science products cannot keep pace with fisheries critics, elevating the risk of politicization of fisheries decision-making through "advocacy science" and proliferation of "faith-based" solutions (Hilborn 2006; Cowan et al. 2010).

In this paper we develop the argument that to effectively counter politicization of fisheries decision-making it is necessary to clarify when these new "ecosystem-based" approaches to fisheries assessment and management should be viewed as an emergent property of innovation in science and policy, and when they require significant changes in fisheries governance. This requires confronting both the implicit and explicit distinction between the EAF and EBM. Although different fisheries regulatory agencies have adopted one or the other of these terms, for reasons that are rarely specified, there is an important difference between the concepts underlying the two terms. The notion of adopting an "ecosystem approach" is inherently evolutionary. Existing fisheries institutions and practices are taken as the starting point, and their policies and techniques (both assessment and management) are augmented with key ecosystem components as these become relevant and feasible. The notion of adopting an ecosystem *basis* for management (EBM) is more revolutionary. One starts with as full an understanding of the ecosystem structure and function as possible, and then decides how human uses, including fishing, can be accommodated without serious alteration to ecosystem processes. Both concepts have strengths and weaknesses. The EAF builds on what exists, requiring fewer changes to legislation and institutions. It focuses efforts on the ecosystem factors considered most relevant to fisheries and can advance as rapidly as science can provide the empirical support for change. However, simply by being incremental, it allows agencies to avoid hard problems both in accommodating less tractable ecosystem relationships and drivers in assessment and management, and in integrating fisheries with other human uses of the services from the same ecosystem. The EBM immediately requires confronting the drivers and relationships that matter most to the ecosystem, whether tractable for science and management to address or not, and requires an integrated approach to decision-making across human uses of ecosystem services. Ecosystem-based management requires changes to legislation and regulatory roles for agencies, and some loss of autonomy of fisheries management agencies relative to more integrated planning and policy, a change not welcomed by most fisheries agencies.

Regardless of approach, neither the science of ecosystem assessments nor the policy of integrated ecosystem-scale management is yet mature. Although modest progress has been made (examples include the Convention on Biological Diversity [CBD] 2009; NOAA 2009), consensus has not yet emerged on key components of the scientific basis for ecosystem-scale assessment and management (e.g., the appropriate spatial scale at which management should operate and/or is likely to be effective), nor on the form and extent of integration of fisheries management with other regulatory agencies. However, there is agreement on directions in which past approaches must be altered. As currently practiced, fishery management remains bounded by historical jurisdictional boundaries rather than ecological ones, and progressive science advisors and managers try to fit expanded ecosystem considerations into spatial frameworks that may correspond poorly to functional ecosystem boundaries. Ways that do match practical management scales with scales of ecosystem dynamics must be found and implemented.

A perfect match of ecological and management scales may be impossible, as there is no single spatial scale that will encompass all the key components of the fishery ecosystem. Some of the key benthic communities and habitats may have important structure on scales of tens of kilometers or less, whereas some ecologically important highly migratory predators in many systems disperse and migrate over hemispheric spatial scales, creating linkages among far-distant local food webs.

Detailed ecosystem indicators can be collected at any of these scales, but their use is likely to vary (Link 2010). At local scales, ecosystem indicators can be proposed as triggers in management control rules, although at present ecosystem properties are rarely used to set tactical management advice, other than bycatch limits and management of forage fish fisheries (Pitcher et al. 2009; Link 2010). As management scales get larger, strategic as well as tactical measures may be at least informed by the status of ecosystem properties. For example, in Australia, management does a qualitative, risk-based screening to identify a few particularly high-risk species from a diverse array of marine organisms (Hobday et al. 2011), and then designs management approaches that consider the risk to those species. The European Commission (EC) established Regional Advisory Committees responsible for strategic and some tactical management advice on fisheries involving more than 90,000 vessels using dozens of different types of fishing gears. Even these committees operate on multiple scales, ranging from major seas to larger areas to capture both demersal and highly migratory species (Jennings and Rice 2011).

In this essay, we attempt to identify some of the most significant challenges to the development and implementation of fisheries in its broader ecosystem context. Specifically, we will seek answers to the following five questions: (1) has conventional fisheries management really failed?; (2) can short-comings in conventional fisheries management be augmented with new socioeconomic management measures, such as allocation of rights?; (3) can short-comings in conventional fisheries management be augmented with new ecologically based measures of EAF, or is EBM necessary?; (4) is restoration of degraded ecosystems a necessary component of achieving the broader mandate or is it sufficient to simply to prevent further degradation?; and (5) if the answer to questions 2, 3, or 4 is yes, how does the added complexity of assessment, management, and ecosystem restoration change the pathways and time course of implementation, as well as the expectations of management outcomes? Question 1 sets the context for the rest of the paper and is addressed first. The possible ways to address any problems identified in considering question 1 are discussed from three perspectives relating to the types of challenges that arise from making fisheries management more holistic and inclusive: empirical, jurisdictional, and societal. That discussion provides the basis to answer the other four questions, two of which deal with pathways ahead and two with breadth of accountability for fisheries management.

We note that consensus was not an explicit requirement of authors for participation in this essay, so the reader may feel at times the discussion does not lead to a single clear and consistent conclusion. In fact, if we had achieved consensus, we might have failed to adequately expose the complexity of the relationship between EAF/EBM and the challenges to implementation of the process.

HAS CONVENTIONAL FISHERIES MANAGEMENT REALLY FAILED?

The answer to this question depends on what one considers success. Notwithstanding dramatic claims about the large majority of stocks globally being overfished (Pauly 2007) or destined for economic extinction by the middle of this century (Worm et al. 2006), recent reports (FAO 2009) document that fewer than 20% of fish stocks globally are overfished or collapsed. Also, Worm et al. (2009) and Branch et al. (2011) show that some major ecosystems have never been overfished, that many others are now recovering from historical overfishing, and that this recovery has been accomplished with conventional fisheries management tools: restrictions of catch and effort, gear limitation, closed areas, and rights-based management (also see review by Jensen et al. 2012). However, the data available for these studies do not include those from small stocks targeted by artisanal or recreational fisheries and includes only a few stocks from Asia and Africa. Consequently, even when judging "success" by the status of the target species, the jury is still out, with some clear failures, some likely successes, and many "currently unresolved" cases.

The success of fisheries management must be judged on standards more inclusive than just the status of the target stock(s). Well-managed fisheries must have sustainable outcomes biologically, economically, and socially. Hilborn (2007) indicated that one of the showcases of sustainable biological management, the Bristol Bay salmon fishery, has been far from successful economically. Social scientists, because of the consequences of fishery regulations for fishery-dependent communities, have taken to task many fisheries considered both biological and economic successes (Lowe and Carothers 2008; Charles and Wilson 2009). Societal success can be even more difficult to define than ecological or economic success, even though it has been one of the three cornerstones of sustainability since the concept entered the policy arena in the 1980s (World Commission on Environment [WCED] 1987); many sources (Rice 2009) stress that a key advance to a more expanded mandate for fisheries management is to make the human dimensions of the fishery an inherent part of the approach. This situation brings out one of the real challenges of more inclusive approaches to fisheries: the more holistic the basis for fisheries management, the more opportunities there are for fisheries to be considered "failed" somehow in policy or implementation. This should not be thought of as a fault of broadening the mandate of fisheries; it is correct to evaluate success of fisheries on all three dimensions of sustainability. However, considering humans as part of the ecosystem explicitly illustrates the difficult standard on which success or failure should be objectively measured. To our knowledge, no meta-analysis has been made of the social or economic status of global or even regional fisheries. Therefore "success" of fisheries management remains unresolved, although indications are ample that specific fisheries can improve performance on any or all of the three aspects of sustainability. For the rest of this essay we consider the challenges to making various types of improvements: empirical, jurisdictional and financial, and societal.

EMPIRICAL CHALLENGES

Although the expanded mandate places fisheries management in a fuller ecosystem context, fisheries management is still management of *fisheries*, not of ecosystems, just as classical fisheries management is management of *fisheries*, not of target species per se. So it is possible to argue that the scales at which fisheries are defined by the coupled natural–human systems (Liu et al. 2007) also define the appropriate scales for management in any context, conventional or ecosystemic. However, coupled natural–human systems are often difficult to delineate (Liu et al. 2007).

Legal jurisdictions always influence how coherent fisheries are defined; for example, vessels cannot operate outside the geographic areas for which they are licensed to operate. This is true even on the high seas, given flag–state responsibilities. Fishers also at least generally respect the legal boundaries of the jurisdictions for which they are licensed to fish, if only because the penalties for unreported and unregulated fishing is severe in an increasing number of areas (FAO 2002a,b).

Beyond the legal jurisdictions, coherent management of fisheries has to be compatible with the distribution and behavior of target species of fisheries. Even in multispecies fisheries, fleets will fish where they expect favorable catch rates and low fishing costs and where the mix of species will keep them at least close to the allowable harvests for each species in the complex being harvested (Alverson and Hughes 1996; Broadhurst et al. 1997; Diamond 2004; Cotter et al. 2009; Piet et al. 2009). Hence fishers' behavior is both a constraint on what options are worth discussing and also an opportunity to focus management on scales that match management measures to the inherent scales of the activities being managed. Taking advantage of that opportunity to focus management on the scales of activities being managed is to select the suites of measures that encourage fisheries to behave in biologically sustainable (for the target species and ecosystem impacts on habitats, bycatch, etc), economically sustainable (likely to have high catch rates and low costs to fish), and socially sustainable (provide livelihoods to those dependent on the fishery) ways. Selection of those suites of measures requires considering the scales of dynamics of the whole ecosystem, going well beyond the communities of the fishers.

Some of the factors needed to define appropriate scales for a greater ecosystem context for management are predictions of large-scale (decades in time, 1000s of kilometers in distance) oceanographic and climatic processes (North Atlantic Oscillation, Pacific Decadal Oscillation, location and paths of boundary currents, upwelling zones, etc.), while others are intermediate and local-scale features in time and space (bathymetry, types and distributions of sea floor substrates, upwelling sites, jets, estuarine plumes, etc.). Factors will differ according to target species, even in the same general geographic management area (small pelagics, large pelagics, demersals, salmonids, etc.), as individual fishers often participate in multiple fisheries, each one for a single or small suite of ecologically similar target species.

Trophodynamic (food web) factors also have to be taken into account in defining the appropriate scales for an expanded management mandate. Again, however, the scales of relevant trophodynamic processes often depend on the characteristics of the target species of the fishery being managed rather than on some general ecosystem property. For example, the trophodynamic scale of considerations for management of the Canadian Pacific sardine fishery must include the entire California Current, down to the southern Baja Peninsula, whereas the trophodynamic scale of considerations for the Canadian Pacific hook-and-line rockfish fishery is much smaller, probably never more than a few hundred km along the coast, and often not more than a few tens of km.

Inclusion of the socioeconomic dimensions in management does not make identification of the appropriate scale for management any easier. The behavior of a group of fishers can be influenced simultaneously by long-standing cultural norms of a local community, the changing preferences for eco-certified fish in distant markets, and fuel prices, which are driven by global geopolitics (Kaplan and McCay 2004; Hutton et al. 2008; Ommer et al. 2009). Ignoring any of these scales in policy or management can lead to bad choices on the social, economic, and even ecological dimensions of sustainability (Degnbol and McCay 2007; Murray et al. 2008).

Consequently, the search for the correct or appropriate scale for management to meet an expanded ecosystem mandate may be futile and misguided. Progress can be made on goals for both target stocks and ecosystems as long as ecosystem considerations are taken into account at whatever scale makes sense for a particular fishery and the stocks it takes as harvest or bycatch. Knowledge of both the main target species supporting each fishery, and the economy and culture of the communities where the fishery is based, are needed to select a sound starting point for examining which considerations (probably on a variety of scales, from basinwide to local) will have to be taken into account in managing a particular fishery. This will be the case whether a fishery is being managed solely in a sectoral context (EAF) or in a broader context integrated with uses of other ecosystem services (EBM).

Because of the many trade-offs necessary to address the ecological, social, and economic considerations described above (also see review by Link 2010), clearly no single operational application for making management more ecologically inclusive will be universally appropriate. Rather, applications fall into a continuum of approaches, delineated in part by their complexity. At one end of the continuum, some of the less complex (but not simple) views of the necessary expansion of management focus on giving greater consideration to the effects of environmental drivers on stock dynamics (Bailey et al. 2004; Stige et al. 2006) and/or to the full footprint of the fishery on the ecosystem (Jennings and Kaiser 1998; Hall 1999). Simply put, expanding the scope of management means incorporating ecosystem principles (sensu Pikitch et al. [2004], Francis et al. [2007], and Link [2010]) into the management of fisheries, so that "success" means maintaining population sizes large enough that exploited stocks can retain their ecological functions, while protecting habitats and critical species.

At the other end of the continuum, more inclusive (and complex) views posit that modern management must explicitly consider whole ecosystems, the fisheries that are imbedded within them, and the cumulative impacts of other pressures on the ecosystems being exploited. This view takes the perspective that marine ecosystems comprise a myriad of interacting species, some targeted by fisheries and some not, located in a spatial domain that contains a variety of habitats and physical oceanographic processes. These ecosystems provide many goods and services to humans and are affected directly and indirectly by many human uses. To an extent great enough to require integrated planning, the species, habitats, and physical processes are interdependent; the uses and impacts are cumulative and interact; and all of these relationships must be taken into account in management lest ecosystem functions diminish.

The more inclusive approaches are certainly closer to views that fisheries management should focus on preserving the full structure, productivity, diversity, and resilience of marine ecosystems (Jackson et al. 2001; Garcia et al. 2003; Browman et al. 2004), not just take account of the environmental forcers that may drive the dynamics of the stocks being harvested and the key direct impacts of the fishery. This may be especially important in large river deltaic ecosystems, estuaries, reefs, shallow continental shelves, etc., where multiple stressors, including climate change, contribute to habitat loss and/or destruction, eutrophication, hypoxia/anoxia, and pollution; such stressors threaten the ability of these areas to provide future ecosystem services, including productivity of fisheries. It necessarily follows, though, that this view of fisheries management makes fisheries just a part of integrated management. Choices to promote sustainable fisheries and efforts to restore productivity of impacted systems in these multistressed ecosystems must necessarily be harmonized with management of the environmental stressors as well.

This more inclusive view of fisheries management also requires a finer understanding of spatial patterns and dynamics, ecosystem-level processes and feedbacks, and the potential consequences of unintended removals and other effects on the ecosystem (Browman et al. 2004; Pikitch et al. 2004). Consequently, ecosystem processes define scales necessary for assessment and management even if they do not coincide with the scale on which fisheries are prosecuted.

Empirical demands differ at different points along the continuum. More fishery-centered approaches may be more practical for tactical advice on the hard decisions about the necessary trade-offs on the ecological, social, and economic dimensions of a specific fishery. At this scale, relatively focused assessments give priority to the ecosystem factors that are significant drivers of the dynamics of the harvested stocks and directly affect the fishery. Assessments to support this scale of approach are not simple but often are tractable. Scales of assessment and management can be matched to scales of fisheries. More fishery-centered approaches also use existing fisheries governance structures to make fisheries decisions, often an important consideration in getting buy-in from the fishers whose activities are being managed and in meeting requirements of fisheries legislation. However, this approach is sectorally based, rather than set up to integrate other human uses of the same place, and may not be consider many key aspects of ecosystem structure and function directly.

The more inclusive end of the continuum is certainly more "ecosystem-oriented" but also is much less tractable. Its best uses may be in developing strategic advice, when there is a need to evaluate restoration options and alternative multiple-use scenarios in systems not yet degraded and to consider how these alternatives are predicted to affect fisheries resources in the future. In this case, scenario evaluations, including social and economic trade-offs over longer time scales, will probably depend more on the growing science of ecological forecasting and policy gaming than on more focused fishery-based assessments. Although many examples in the recent literature argue for and attempt to apply a more holistic approach, all acknowledge the empirical challenge of ecosystem-scale assessment of status, and most note that more holistic assessments often have to stop several steps away from providing an actual basis for tactical decision-making in any particular fishery (NRC 2006; Bakun et al. 2009; Game et al. 2009; Gerber et al. 2009; Hobday et al. 2009; Lindegren et al. 2009; Litzow and Urban 2009; Morell 2009).

Substantial progress has been made in the development of ecosystem-scale assessment tools that are capable of integrating physical forcing with dynamics of populations or functional groups, which often include the pressures on the system from multiple human activities in the sea (Travers et al. 2007; Link 2010; Fulton et al. 2011). Different tools take qualitative ("traffic light"; Caddy 2007) and quantitative (structural analyses of interaction matrices; Robinson et al. 2008; Kenny et al. 2009) approaches to the complex integration of tasks. For the initiatives most closely linked to policy, these ecosystem assessment approaches focus on identifying those ecosystem components most crucial for maintaining ecosystem structure and functions: the "ecologically and biologically significant area/species" of CBD (2009) and DFO (2006), and the "vulnerable marine ecosystems" of United Nations General Assembly Resolution 61/105 (UN 2006) and FAO (2009). Although these approaches can clarify the nature of some of the major trade-offs that must be made among human uses (including among different fisheries) and between ecological, social, and economic aspects of policy choices, they are at present mostly strategic tools. They usually fall short of providing an analytical basis for determining quantitative ecological benchmarks and triggers for decision rules.

Many quantitative descriptive and simulation models are now available, ranging from expanded single-species assessments that consider changes in predation mortality, to so-called "end-to-end" models that attempt to capture both the biological and biophysical processes governing production dynamics (Fulton et al. 2011). Plagányi (2007) and Espinoza-Tenorio et al. (2012) provide thorough overviews of the strengths and limitations of each model platform, and despite tremendous advancement in this field over the past decade, there will always be limitations in forecasting skill from these models because of limitations in our fundamental knowledge of ecosystem dynamics and structure. In fact, some argue that continued use of assessment and ecosystem models for management decisions should be heavily subsidized by large-scale monitoring of fish abundance and CPUE (i.e., Large Marine Ecosystems; Branch et al. 2011; Hilborn 2012) if we are to sustain fisheries worldwide, despite the high costs of collecting data at very large spatial scales.

Clearly, empirical challenges will be encountered during assessment of ecosystem status at every scale of management. At tactical scales, existing assessment algorithms will have to be augmented by more functional relationships between major ecosystem forcers and stock dynamics, assessment of direct impacts on parts of the ecosystem other than the target species, and more attention to the social and economic consequences of management options. For strategic management at the ecosystem scale, models and assessment tools that are much more inclusive will be needed, considering species interactions, "significant" areas and species, and interactions between the effects of different industry sectors. Progress can certainly be made on fishery and ecosystem objectives when ecosystem-based principles are applied at relevant scales. However, many years will be needed before it will be clear what efforts will be needed, and at what scales, to truly achieve the goals of preserving the structure and function of marine ecosystems, allowing coastal communities to maintain their preferred livelihoods, and having fisheries create wealth rather than require support for continued existence.

Addressing the challenges of more inclusive ecosystem assessments will require more than just larger data sets with more variables. A history of exploitation has already affected the dynamics of many stocks and ecosystem components (see review by Jensen et al. 2012). Essentially all population dynamics models are strongly influenced by the density-dependent feedbacks structured into the models. Multispecies trophodynamic models create more ways in which density dependence can be expressed, but do not change the dominant role of these feedback terms in the behavior of the modeled systems (Pope et al. 2006). If most data come from a time when fishing has already reduced abundances of species and functional groups to well below ecosystem-scale carrying capacities, the data may provide only limited insight into how these feedbacks are expressed. Community dynamics are undoubtedly still regulated, but not by the functional processes that would have regulated them when most large fish were near their respective carrying capacities.

Aside from altering the position of a stock's spawning biomass relative to its carrying capacity, the historical pattern as well as intensity of fishing can also affect population dynamics processes in another way. Many marine fish spawn in the pelagic realm and rely on ocean currents to transport their eggs to a diverse array of larval and juvenile development/nursery grounds (Houde 1987, 1989, 2008; Bakun 1989; Cushing 1996). It has been argued that this can be viewed as an insurance policy, because at the time of spawning, a fish cannot know which habitat(s) will be optimal in any given year. If abundance is reduced, particularly if metapopulations have been depleted, the population may fail to saturate all potentially suitable habitats. This can result in greater interannual variability in recruitment of the population, and increases in the vulnerability of the recruits to local habitat degradation, if the degraded habitats are the ones most often used by the remaining stock (NRC 2002). In such conditions, the study of recruitment ceases to be an attempt to estimate the stock–recruit curve, but also requires understanding how processes work across life stages (Houde 2008). It also requires a bigger research agenda than has been undertaken historically.

Despite these (and many other) complexities, a great deal of knowledge is now available that can be used in policy and management (Murawski 2007). Moreover, not every ecosystem process and relationship needs to be quantified to make significant progress in these areas.

Even at the ecosystem scale, management still can apply a large dose of common sense, building on the obvious truth that fewer, smaller impacts are better than many large ones. As the NRC (1999) report on Sustaining Marine Fisheries noted, the first step in moving fisheries management towards incorporating ecosystem considerations is good single-species management. Research projects around the world are studying ecosystems under stress from fishing and other causes (e.g., Halpern et al. 2008). Research is increasingly consisting of collaborations between social scientists, economists, and ecologists to better understand the root causes of overfishing and to develop decision-support tools that close the gap in moving from single-species management to more ecosystem-oriented approaches.

However, the sheer complexity of multiple life stage trophic and habitat interactions is sure to create many surprising and highly counterintuitive dynamic responses to any ecosystem policy initiative (Pine et al. 2009). Adding the human dimension of the ecosystem approach will provide many lessons to ecologists on why supercilious platitudes of "just take less" have been unhelpful in the classical single-species approach and will continue to be unhelpful in an ecosystem context. There is great value in treating each management initiative as a deliberate, adaptive management experiment with highly uncertain outcomes (Walters 1986). However, the application of experimental adaptive management approaches in fisheries has so far been extremely limited (Jensen et al. 2012). Severe problems, ranging from inadequate institutional incentives to excessive monitoring costs, have plagued these approaches (Walters 2007), and policy-makers often show some resistance to experiments with people's lives and livelihoods, no matter how much their ecological advisors tell them could be learned from the results.

JURISDICTIONAL AND FINANCIAL CHALLENGES

Another important issue that will scale directly with the scale and scope of management objectives and measures is the distribution of wealth and the jurisdictional authority to act. To the extent that management decisions can be made and implemented locally, these issues may not be much greater than they are in local single-species management. However, the larger the list of ecosystem factors that must be considered in a decision or management action, the less likely it is that decisions can be made locally and by a single regulatory agency. Even if those prosecuting a fishery experience it as local, key drivers may be acting elsewhere or on larger scales, and important impacts may affect fishers or ecosystems in other areas or users of other ocean resources. Hence decisions about ecosystem-sustainable options for a local fishery may require consideration and assessments on much larger scales. Costs of measures to improve the profitability or ecological sustainability of the fishery many have to be borne by others, often not fishers, who may not share in subsequent benefits. The agencies with authority to implement the necessary measures may not be the local fishery agencies, or possibly may not be fisheries agencies at all. More inclusive and holistic approaches are not going to make life easier for any agency or user of any ocean resource.

One thing is certain; it costs money to collect data and manage fisheries. For example, there is correlation between the number of recognized fisheries collapses (and recoveries) and the relative wealth of the countries reporting these findings (Figure 1; Worm et al. 2009). This correlation is not perfect, as some wealthy countries apparently are better than others at managing fisheries; moreover, collapses occur more frequently in eastern boundary currents (upwelling areas) that experience decadal-scale replacements of species complexes even when there is little or no fishing (Roy et al. 1989; Lluch-Belda et al. 1991; Santos et al. 2001). On the other hand, this correlation occurs in part because the developed world can afford to capitalize large-scale fisheries, and in part because wealthy countries can afford to collect and manage the fisheries they prosecute (Worm et al. 2009).

One good example of the disparity in relative wealth, jurisdictional conflicts, and their effects on fisheries management at the large marine ecosystem (LME) spatial scale is the pan-Caribbean basin, including the Gulf of Mexico, which is bounded by the USA, Mexico, and Cuba (Figure 2). The number of collapsed (and recovered) fisheries, from among the 72 now prosecuted by the USA in the Gulf, is relatively high and diverse. These include highly migratory species and reef fishes that occur in waters of all three countries. Along with the obvious disparity in wealth, there is an associated asymmetrical distribution of landings data, with scarce and significantly poorer data quality being obtained from poorer countries. This has forced the USA to assess shared stocks in the Gulf of Mexico LME on the basis of only US landings data. Even for single-species management, this approach has been problematic because we know that significant landings of some shared species are made in Mexico and Cuba with little or no regard to ecosystem-based considerations, particularly regarding bycatch and size limits (Rose and Cowan 2003).

Jurisdictional challenges also can arise over large spatial scales of management even when all partners fully intend to collaborate and cooperate to achieve common goals. Many jurisdictions are developing inclusive advisory or management

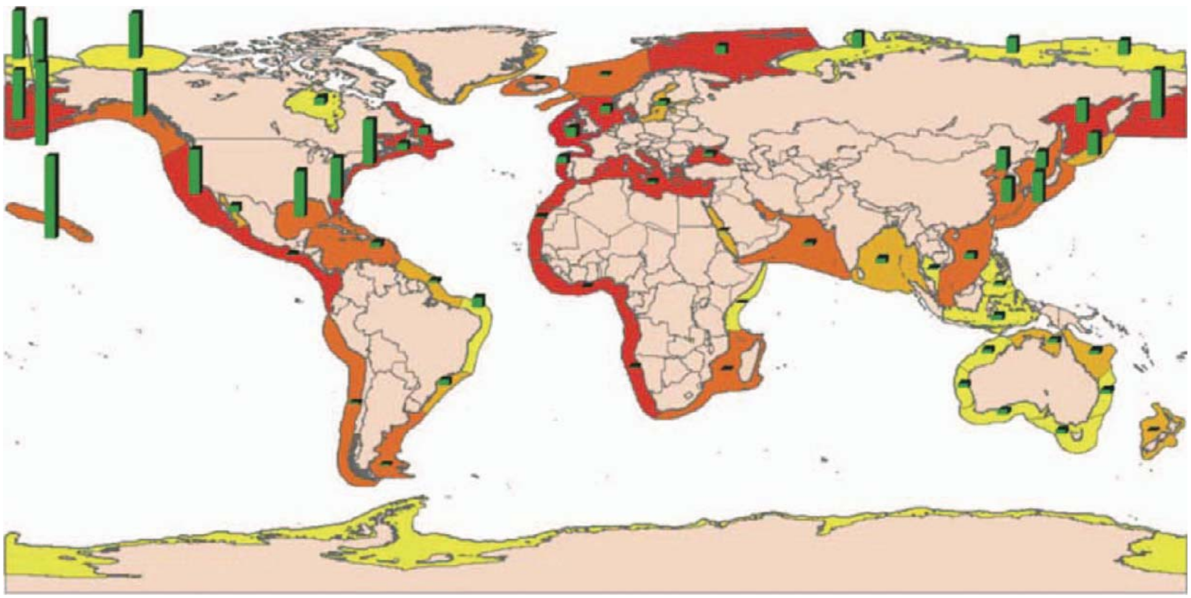

FIGURE 1. Disparity in purchasing power in countries that manage fisheries in one or more of the world's LMEs. The size of countries' economies has been converted into international dollars (also known as the Geary–Khamis dollar, a hypothetical unit of currency that has the same purchasing power of the U.S. dollar in the USA at a given point in time) using purchasing power parity (http://fx.sauder.ubc.ca/PPP.html). Green bars represent millions in 2012 international dollars. Filled areas represent collapsed fisheries on a color gradient of yellow to red. Yellow represents 0–60 collapses while red represents 345–544 collapses.

councils such as the US Regional Management Councils, the European Union (EU) Regional Advisory Councils, and the Australia Fisheries Management Authority. Their decision-making powers and detailed modes of operation differ, but all include a spectrum of perspectives from fishers to conservation interests and are intended to make fisheries decision-making more representative and balanced. However, the appropriateness of having a broadly inclusive process for provision of management advice is part of the original Code of Conduct for Responsible Fishing, and was a good idea long before ecosystem approaches were acknowledged as central to good management. Moreover, although all of these national/regional advisory bodies have at least illustrative cases of making fisheries management consider larger ecosystem perspectives, all are perceived by other agencies as having made fisheries management even more sectorally entrenched and less integrated with decision-making of other regulatory agencies.

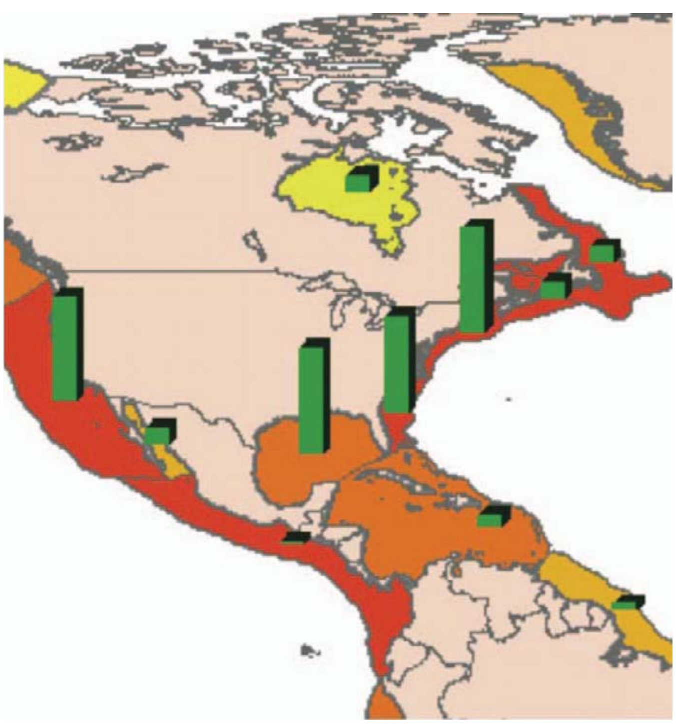

FIGURE 2. Disparity in purchasing power in countries that comprise the Gulf of Mexico LME. See Figure 1 for details.

SOCIETAL CHALLENGES—PRESENT AND FUTURE

Society will ultimately determine the future approaches taken for fisheries policy and management and the scopes and scales at which they will be implemented. Even in today's world, society is finding it hard to make and stick to clear decisions about the direction desired for fisheries management and policy. Even among the authors, there is little consensus on the "right" outcome ("targets") at which policy and management should aim. However, there is general agreement on a few important points that bound the debate about selecting policy and management targets:

- it is necessary and feasible to manage the ecosystem state to some extent, at least regarding the direct impacts of fisheries (and other human activities),

- it is impossible to manage ecosystems entirely: too many drivers are not under human control and the ability to predict ecosystem responses to perturbations (or management interventions) is still limited.

Between those boundaries there is ample scope for societal—and expert—debate on the degree and nature of impacts to accept and on the accountability of managers for the ecosystem effects of fishing (Jensen et al. 2012). Large-scale, intensive aquaculture alters aquatic systems in ways that approach the agricultural model. Such practices can effectively address pressing food security issues for protein-poor regions of the world, but the effects on the ecosystem are large and strongly contested (Rice 2011). Efforts to enhance the natural productivity of depleted wild stocks can range from habitat improvements to active augmentation of recruitment. Agreement is almost universal that inclusive approaches are needed to restore degraded ecosystems, and that controlling sources of pollution, nutrient enrichment, direct habitat destruction, etc., is feasible (Ring et al. 2010; Seabrook et al. 2011). However, when it comes to restoring the structure of biotic communities, there are serious reservations about the ability of humans to "steer" degraded ecosystems towards states that resemble historical conditions (Jackson et al. 2001; Cowan et al. 2008).

For most marine and coastal systems, fishing has already altered the ecosystems sufficiently that setting appropriate targets cannot be informed by the pristine structure and functioning of an ecosystem, and there is debate about even what natural ecosystem processes would mean (Ring et al. 2010; Seabrook et al. 2011). However, new calls are being made to use fisheries management successes and failures as ecological experiments, to be mined for the information they contain about sustainability (Jensen et al. 2012).

Some examples of "back to the future" approaches do exist. In the USA, most notable are past and ongoing efforts to restore wetlands and ecosystem functioning to enhance fisheries productivity in the San Francisco Bay-Delta (California Code Chapter 3: San Francisco Bay Restoration Authority [66702.–66702.5]). Hundreds of millions of US dollars have already been spent to develop a real-time monitoring system capable of limiting the intake of endangered species in pumps that deliver water to southern California. More importantly, management of this system requires that the estuarine null zone be maintained within defined distances downstream to ensure that sufficient flows exist in the estuary to promote recruitment success of estuary-dependent fishes and invertebrates (Interagency Ecological Program; http://www.iep.ca.gov/pwt.html; CALFED Bay-Delta Program, http://www.science.calwater.ca.gov/science_index.html/). Similarly, restoration efforts in Chesapeake Bay are focused on reducing nutrient inputs and improving other ecosystem indicators to specified benchmarks to limit eutrophication and the development of bottom water hypoxia/anoxia. Among the most important reasons given for the many actions already taken is to ensure that the Bay is restored to a point where it is capable of supporting future living resources and fisheries productivity at levels similar to the historical past (Chesapeake Bay Executive Order 13508, November 2009, http://executiveorder.chesapeakebay.net/).

Other notable examples include (1) watershed restoration in the Pacific northwest, a costly ecosystem management/restoration program in the USA and Canada aimed explicitly at insuring healthy water systems for Pacific salmon (Nehlsen et al. 1991; Pacific Salmon Restoration Program, http://www.ifrfish.org/pacificsalmon; http://www.wsu.edu/swwrc/SalmonIndex.html), and (2) water management and restoration of the Florida Everglades and Florida Bay to promote spiny lobster and others fisheries (Comprehensive Everglades Restoration Plan, http://www.evergladesplan.org/index.aspx). Similar large-scale restoration and management programs are planned for the Great Lakes (US EPA Great Lakes Restoration Plan, http://www.epa.gov/grtlakes/glri/glmyrapo.pdf) and northern Gulf of Mexico (Mississippi River deltaic ecosystem–Coast 2050 [LCWCRTF 1998]; CPRA 2007), both of which include specific goals aimed at restoring or maintaining fisheries productivity.

All these major restoration programs are aimed primarily at reducing and repairing habitat loss, with restored fishery productivity as one of many goals. However, even when there was consensus on goals to achieve improved ecosystem status of these degraded systems, ecology alone could not identify which of many restored states was the best (or at least most acceptable) goal. In addition, few of the restoration measures are applied by, or under the direct control, of fisheries managers and policy makers. The important message is that in situations where ecosystem restoration is necessary for sustainable fisheries, integrated planning and decision-making across many industry and societal sectors is essential both to ensure coordinated action among agencies responsible for managing the different stressors on habitat quality and fisheries productivity, and for reaching consensus and resolving conflicts among agencies and civil society on the major trade-offs needed for restoration. The targets at which to aim will be compromises of many social and economic considerations as well as ecological ones arrived at through processes inclusive of many parties beyond fishers and fisheries managers.

We would be remiss if we failed to mention marine protected areas (MPAs) and other related types of spatial management approaches in the tactical context of recovery/restoration and fisheries management more generally. Although many goals of MPAs are the same as those of fisheries management, there are some important differences and the literature has considered the linkage between MPAs and fisheries management as anything from independent to one and the same (Halpern et al. 2010). We do not believe that MPAs alone are sufficient to achieve all of the objectives of fisheries management because they rarely are large enough to generate more than very local benefits to fisheries (Defeo and Perez-Castaneda 2003; Martell et al. 2005; Hilborn 2006; Laurel and Bradbury 2006); and to meet basic criteria for being an MPA, when in conflict, conservation goals *must* be given priority over any social or economic goals. MPAs large enough to achieve all the ecological objectives, even restoration, would require such significant redefinitions of the social and economic objectives of the fisheries that at least some of the participants in the fisheries would argue the management regime was no longer a fisheries regime but a protected-areas regime.

In the future we expect these challenges to grow larger, not smaller. For narrow scopes and tactical scales, decisions may be driven by the same incentives that have driven conventional fisheries management, namely, short-term, fitness-based incentives. Incentives can be defined by whatever metrics one chooses (economic, nutritional, etc.), but keep in mind that the social culture making the decisions evolved during a time of ecological abundance.

This is relevant here because Day et al. (2007, 2012) showed that fishery yields in estuaries during the last postglacial maximum in sea level are 10-fold higher than in open marine ecosystems, and that the distribution of complex, urban societies first formed in the presence of abundant coastal resources associated with salt marshes and mangroves (Figure 3). Abundant resources freed humans to think beyond sustenance, thus creating luxury time to develop language, arts, culture, and religion, all of which are believed to have benefited humans during their rapid, recent evolution (Dennett 2006; Day et al. 2007, 2012; Hall and Day 2009). It is also clear that evolution in the time of ecological abundance promoted short-term thinking wherein humans have used the experiences of the past two or three generations to inform decisions about what do for the next one or two generations (Sagan and Druyan 1992; Wilson 1998; A. Einstein, cited in Isaacson 2007).

In contrast, we are now facing ecology in time of scarcity (Hall and Day 2009), and the history of human activity in the world's oceans extends well beyond overfishing (Jackson et al. 2001). Populations in traditionally fishery-dependent coastal communities are dwindling as members of the younger generation fail to follow their fathers and forefathers into fisheries (Hamilton and Otterstad 1998; Hansen 1999; Tietze et al. 2000). The forecasted negative impacts of climate change on crop and livestock production in many areas that historically have provided at least subsistence if not surplus food (Gregory et al. 2005), combined with projected growth of human population that is also increasingly urbanized in large coastal cities, is creating a threat of a food security crisis that must be met in large part by food from aquatic (freshwater and marine) ecosystems (Cochrane et al. 2009; Hall and Day 2009). We may be living at a nexus when not just the ecological basis for decision-making about fishery resources is changing fundamentally, but

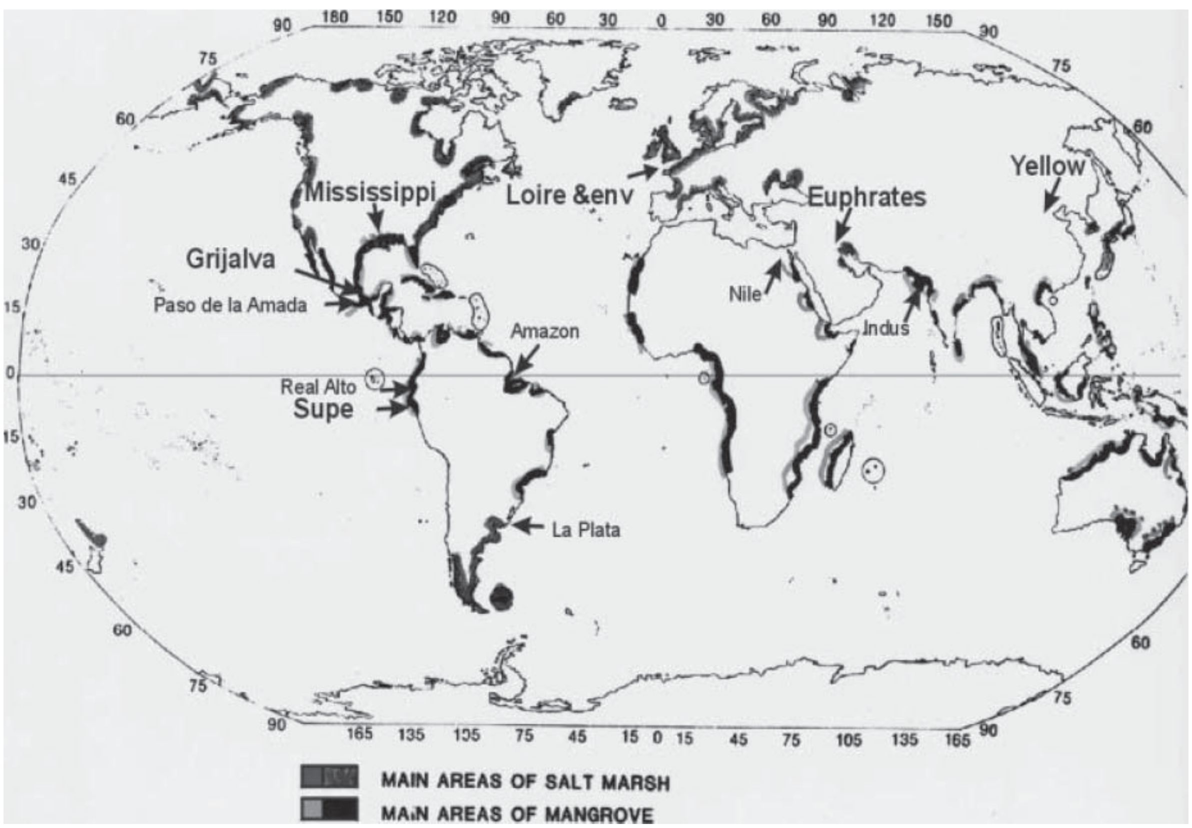

FIGURE 3. Distribution of early urban societies. The distribution of coastal mangroves and salt marshes is indicated by dark and light shades (adapted from Day et al. 2012).

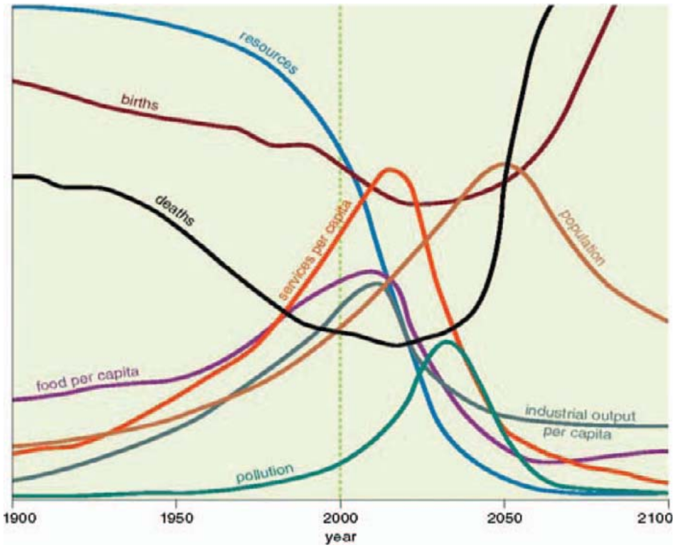

FIGURE 4. The original projections of the limits-to-growth model (Hubbert 1969) examined the relation of a growing population to resources and pollution but did not include a timescale between 1900 and 2100. Hall and Day (2009) projected forward from the halfway mark of 2000; the projections up to the current time are largely accurate, although projections for upcoming years predict wild oscillations.

also in a time when the economic and social/cultural basis must change.

As such, societal challenges may be the biggest challenge to expanding fisheries to a broader ecosystem context because we question whether changes can occur quickly enough for society to fully embrace these concepts and measures, especially in ecosystems where restoration is needed to ensure future fisheries productivity. Restoration often cannot be accomplished quickly, and it frequently requires a great deal of wealth to implement, particularly when top-down anthropogenic stressors have resulted in punctuated equilibria and shifts in ecological baselines (Jones and Walters 1976; Jackson et al. 2001). We are not certain that the history of human evolution has prepared society to understand and tolerate the trade-offs needed to forego short-term economic benefits and to abandon open-access utilization of natural resources in favor of protecting or restoring ecosystem goods and service over much longer time periods, especially in times of ecological scarcity (Figure 4; Hall and Day 2009).

The costs of such actions will continue to increase in times of scarcity as the price of fossil fuels climbs in response to diminishing supply and potable water becomes scarce, yet ecosystem restoration can take decades to show any net benefits and centuries to complete. The most visible example of this dilemma is the ongoing debate about climate change, despite the overwhelming evidence that it is occurring and having large consequences (Pachauri and Reisinger 2007). For EAF to succeed, even if the goal is simply to draw a line in the sand and prevent further degradation of ecosystems as they exist today, it probably will be necessary to begin forward-thinking over time scales longer than have been necessary in the past. If the updated projections of the Hubert model in Figure 4 are even modestly realistic, relatively rapid future declines in food per capita and in ecological goods and services will make these challenges more pressing, thus requiring actions sooner than later.

HAVE WE ANSWERED THE QUESTIONS WE POSED?

Question 1.—Has conventional fisheries management really failed? At some points in their past, many stocks have been overfished, and in some jurisdictions and periods, overfishing was severe and widespread. Nonetheless, some major ecosystems have never been overfished and many others are now recovering from historical overfishing. So conventional fisheries management has not categorically failed, but neither has it uniformly succeeded, relative to the traditional stock- or fishery-specific objectives for management. Moreover, complete data are available only for some larger-scale fisheries of the world. Adequate information from most of the world's artisanal fisheries and many large-scale fisheries in the Southern Hemisphere are not available for analysis, so a full evaluation of success is not yet possible.

Question 2.—Can shortcomings in conventional fisheries management be augmented with new socioeconomic management measures, such as allocation of rights? Where overfishing has not occurred and where recoveries are being accomplished, these outcomes are being achieved with conventional fisheries management tools such as catch and effort restrictions, gear limitation, closed areas, and rights-based management. However, recent international agreements have committed fisheries to achieve not just sustainable use of target stocks but indeed sustainable use of marine ecosystems. Some conventional tools may contribute to achieving this broader mandate, closed areas being a favorite tool of marine conservation biologists. Nonetheless, the broader mandate cannot be delivered by fisheries management agencies acting solely by sector and using only the conventional tools of fisheries management in their classical applications. For example, the policy and practice of allocation of rights to targeted stocks is complex, and the allocation of rights to broader ecosystem impacts of fisheries is still at the conceptual stage for all but special cases such as some bycatch allocations.

Question 3.—Can shortcomings in conventional fisheries management be augmented with new ecologically based measures such as EAF or EBM? Considering EAF first, taking account of more drivers of stock dynamics and the broader footprint of fisheries may contribute modestly to achieving the stock-related objectives of conventional fisheries management. It will do little, however, to address shortcomings in conventional management that stem from unresolved conflicts between social, economic, and ecological outcomes of fisheries, or from ineffective implementation of conventional management measures. To the extent that the ecosystem impacts of fisheries are managed effectively, EAF can contribute to the broader mandate of sustainable use of ecosystems. However, as soon as fishery impacts on ecosystems have to be harmonized with the impacts or desired outcomes of other uses of marine ecosystems, EAF alone will not be adequate. Some form of integrated planning and decision-making becomes necessary, moving into the domain of EBM. An ecosystem approach to fisheries has more tractable information demands for implementation, is more readily focused on priority problems, and is more compatible with fisheries governance. However, it fails to address some key determinants of sustainability. Ecosystem-based management deals more adequately with the drivers of unsustainability and the new commitments to ecosystem-scale sustainability. But it demands far more action by policy-makers and managers, more attention to the human dimension of an ecosystem, and an integration of fisheries management and decision-making with management and decision-making of other sectors—an integration so far widely resisted by fishery interests.

Question 4.—Is restoration of degraded ecosystems a necessary fisheries management objective, or is our goal simply to prevent further degradation? If the goal of management is preserving the structure, productivity, diversity, and resilience of marine ecosystems and prosperous, vibrant human communities, then restoration will often be necessary. Restoration is most urgently needed in places where significant habitat degradation has occurred. However, such degradation is rarely due to fisheries alone, even though fisheries productivity is impaired by the degradation, and conventional fisheries management objectives are not achievable until restoration has occurred. Under such conditions, EBM, integration of efforts to manage all the causes of ecosystem degradation, will be needed.

Question 5.—If the answer to questions 2, 3, or 4 is yes, how does the added complexity of assessment, management, and ecosystem restoration change the pathways and time course of implementation and the expectations of management outcomes? Where restoration is needed (question 4), it is rarely accomplished quickly and often requires a great deal of wealth to implement. Where ecosystem-scale sustainability of impacts requires major reductions in the impacts of fisheries on populations or habitats, there will be social and economic consequences of the necessary management measures. Again, more integrated consideration of livelihoods rather than just fisheries' profits and employment will probably be needed before progress is possible. We are not certain that the history of human evolution has prepared society to understand and tolerate the trade-offs needed to forego short-term economic benefits, and abandon open-access utilization of natural resources, in favor of protecting or restoring ecosystem goods and service over much longer time periods.

REFERENCES

Alverson, D. L., and S. E. Hughes. 1996. Bycatch: from emotion to effective natural resource management. Reviews in Fish Biology and Fisheries 6:443–462.

Bailey, K. M., A. B. Hollowed, and W. S. Wooster. 2004. Complexity of marine fisheries dynamics and climate interactions in the northeast Pacific Ocean. Pages 147–151 *in* N. C. Stenseth, G. Ottersen, J. Hurrell, and A. Belgrano, editors. Marine ecosystems and climate variation: the North Atlantic. Oxford University Press, Oxford, UK.

Bakun, A. 1989. L'ocean et la variabilité des populations marines [The ocean and the variability of fish stocks]. Pages 155–187 *in* J. P. Troadec, editor. L'homme et les ecosystemes halieutiques. Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER), Paris.

Bakun, A., E. A. Babcock, and C. Santora. 2009. Regulating a complex adaptive system via its wasp-waist: grappling with ecosystem-based management of the New England herring fishery. ICES Journal of Marine Science 66:1768–1775.

Branch, T. A., O. P. Jensen, D. Ricard, Y. Ye, and R. Hilborn. 2011. Contrasting global trends in marine fishery status obtained from catches and from stock assessments. Conservation Biology 25:777–786. DOI: 10.1111/j.1523-1739.2011.01687.x.

Broadhurst, M. K., S. J. Kennelly, J. W. Watson, and I. K. Workman. 1997. Evaluations of the Nordmøre grid and secondary bycatch-reducing devices (BRDs) in the Hunter River prawn-trawl fishery, Australia. U.S. National Marine Fisheries Service Fishery Bulletin 95:209–218.

Browman, H. I., and K. I. Stergiou. 2005. Politics and socio-economics of ecosystem-based management of marine resources. Marine Ecology Progress Series 300:241–296.

Browman, H. I., K. I. Stergiou, P. M. Cury, R. Hilborn, S. Jennings, H. K. Lotze, P. M. Mace, S. Murawski, D. Pauly, M. Sissenwine, and D. Zeller. 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Marine Ecology Progress Series 274:269–303.

Caddy, J. F. 2007. Marine habitat and cover: their importance for productive coastal fishery resources. UNESCO Publishing, Paris.

CBD (Convention on Biological Diversity). 2009. Expert workshop on scientific and technical aspects relevant to environmental impact assessment in marine areas beyond national jurisdiction. United Nations Environment Programme (UNEP), Report UNEP/CBD/EW-EIAMA/2, New York.

CEC (Commission of the European Communities). 2008. The role of the CFP in implementing an ecosystem approach to marine management: communication from the commission to the council and the European parliament. CEC, COM(2008)187 Final, Brussels.

CEC (Commission of the European Communities). 2009. Green paper: reform of the common fisheries policy. CEC, COM(2009)163 Final, Brussels.

Charles, A. T., and L. Wilson. 2009. Human dimensions of marine protected areas. ICES Journal of Marine Science 66:6–15.

Cochrane, K. L., C. De Young, D. Soto, and T. Bahri, editors. 2009. Climate change implications for fisheries and aquaculture: overview of current scientific knowledge. FAO (Food and Agriculture Organization of the United Nations), FAO Fisheries and Aquaculture Technical Paper 530, Rome.

Cotter, J., P. Petitgas, A. Abella, P. Apostolaki, B. Mesnil, C. Y. Politou, J. Rivoirard, M. J. Rochet, M. T. Spedicato, V. M. Trenkel, and M. Woillez. 2009. Towards an ecosystem approach to fisheries management (EAFM) when trawl surveys provide the main source of information. Aquatic Living Resources 22:243–254.

Cowan, J. H., Jr., C. B. Grimes, W. F. Patterson III, C. J. Walters, A. C. Jones, W. J. Lindberg, D. J. Sheehy, W. E. Pine III, J. E. Powers, M. D. Campbell, K. C. Lindeman, S. L. Diamond, R. Hilborn, H. T. Gibson, and K. A. Rose. 2010. Red snapper management in the Gulf of Mexico: science- or faith-based? Reviews in Fish Biology and Fisheries 21:187–204.

Cowan, J. H., Jr., C. B. Grimes, and R. F. Shaw. 2008. Life history, history, hysteresis, and habitat changes in Louisiana's coastal ecosystem. Bulletin of Marine Science 83:197–215.

CPRA (Coastal Protection and Restoration Authority). 2007. Integrated ecosystem restoration and hurricane protection: Louisiana's comprehensive master plan for a sustainable coast. CPRA of Louisiana, Baton Rouge.

Cushing, D. H. 1996. Towards a science of recruitment in fish populations. Excellence in Ecology 7.

Day, J. W., Jr., J. D. Gunn, W. J. Folan, A. Yáñez-Arancibia, and B. P. Horton. 2007. Emergence of complex societies after sea level stabilized. EOS (Washington, D.C.) 88:169–170.

Day, J. W., Jr., J. D. Gunn, W. J. Folan, A. Yáñez-Arancibia, and B. P. Horton. 2012. The influence of enhanced post-glacial coastal margin productivity on the emergence of complex societies. Journal of Island and Coastal Archaeology 7:23–52.

Defeo, O., and R. Pérez-Castañeda. 2003. Misuse of marine protected areas for fisheries management: the case of Mexico. Fisheries 28(7):35–36.

Degnbol, P., and B. J. McCay. 2007. Unintended and perverse consequences of ignoring linkages in fisheries systems. ICES Journal of Marine Science 64:793–797.

Dennett, D. C. 2006. Breaking the spell: religion as a natural phenomenon. Viking Penguin, New York.

DFO (Department of Fisheries and Oceans Canada). 2006. Total allowable catches for groundfish stocks off Nova Scotia and the Bay of Fundy. DFO, Dartmouth, Nova Scotia.

Diamond, S. L. 2004. Bycatch quotas in the Gulf of Mexico shrimp trawl fishery: can they work? Reviews in Fish Biology and Fisheries 14:207–237.

Espinoza-Tenorio, A., M. Wolff, M. H. Taylor, and I. Espejel. 2012. What model suits ecosystem-based fisheries management? A plea for a structured modeling process. Reviews in Fish Biology and Fisheries 22:81–94.

FAO (Food and Agriculture Organization of the United Nations). 2002a. Implementation of the international plan of action to prevent, deter and eliminate illegal, unreported and unregulated fishing. FAO, FAO Technical Guidelines for Responsible Fisheries 9, Rome.

FAO (Food and Agriculture Organization of the United Nations). 2002b. Report of the Reykjavik conference on responsible fisheries in the marine ecosystem. FAO, FAO Fisheries Report 658, Rome.

FAO (Food and Agriculture Organization of the United Nations). 2003. Fisheries management: 2. the ecosystem approach to fisheries. FAO, FAO Technical Guidelines for Responsible Fisheries 4(Supplement 2), Rome.

FAO (Food and Agriculture Organization of the United Nations). 2009. Report of the technical consultation on international guidelines for the management of deep-sea fisheries in the high seas. FAO, FAO Fisheries and Aquaculture Report 881, Rome.

Francis, R. C., M. A. Hixon, M. E. Clarke, S. A. Murawski, and S. Ralston. 2007. Ten commandments for ecosystem-based fisheries scientists. Fisheries 32:217–233.

Fraser, H. M., S. P. R. Greenstreet, R. J. Fryer, and G. J. Piet. 2008. Mapping spatial variation in demersal fish species diversity and composition in the North Sea: accounting for species- and size-related catchability in survey trawls. ICES Journal of Marine Science 65:531–538.

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.

Game, E. T., H. S. Grantham, A. J. Hobday, R. L. Pressey, A. T. Lombard, L. E. Beckley, K. Gjerde, R. Bustamante, H. P. Possingham, and A. J. Richardson. 2009. Pelagic protected areas: the missing dimension in ocean conservation. Trends in Ecology and Evolution 24:360–369.

Garcia, S. M., and A. T. Charles. 2008. Fishery systems and linkages: implications for science and governance. Ocean and Coastal Management 51:505–527.

Garcia, S. M., A. Zerbi, C. Aliaume, T. Do Chi, and G. Lasserre. 2003. The ecosystem approach to fisheries: issues, terminology, principles, institutional foundations, implementation and outlook. FAO (Food and Agriculture Organization of the United Nations), FAO Fisheries Technical Paper 443, Rome.

Gerber, L. R., L. Morissette, K. Kaschner, and D. Pauly. 2009. Should whales be culled to increase fishery yield? Science 323:880–881.

Gregory, P. J., J. S. I. Ingram, and M. Brklacich. 2005. Climate change and food security. Philosophical Transactions of the Royal Society of London B 360:2139–2148.

Hall, C. A. S., and J. W. Day Jr. 2009. Revisiting the limits to growth after peak oil. American Scientist 97:230–237.

Hall, S. J. 1999. The effects of fishing on marine ecosystems and communities. Blackwell Scientific Publications, Oxford, UK.

Halpern, B. S., S. E. Lester, and K. L. McLeod. 2010. Placing marine protected areas onto the ecosystem-based management seascape. Proceeding of the National Academy of Sciences of the USA 107:18312–18317.

Halpern, B. S., S. Walbridge, K. A. Selkoe, C. V. Kappel, F. Micheli, C. D'Agrosa, J. F. Bruno, K. S. Casey, C. Ebert, H. E. Fox, R. Fujita, D. Heinemann, H. S. Lenihan, E. M. P. Madin, M. T. Perry, E. R. Selig, M. Spalding, R. Steneck, and R. Watson. 2008. A global map of human impact on marine ecosystems. Science 319:948–952.

Hamilton, L., and O. Otterstad. 1998. Demographic change and fisheries dependence in the northern Atlantic. Human Ecology Review 5:16–22.

Hansen, J. C. 1999. Why do young people leave fishing communities in coastal Finnmark, north Norway? Pages 225–251 *in* R. Byron and J. Hutson, editors. Local enterprise on the North Atlantic margin: selected contributions to the fourteenth international seminar on marginal regions. Ashgate, Surrey, UK.

Hilborn, R. 2006. Faith-based fisheries. Fisheries 31:554–555.

Hilborn, R. 2007. Moving to sustainability by learning from successful fisheries. Ambio 36:296–303.

Hilborn, R. 2012. The evolution of quantitative marine fisheries management 1985–2010. Natural Resource Modeling 25:122–144.

Hobday, A. J., S. Griffiths, and T. Ward. 2009. Pelagic fishes and sharks. Page 16 *in* E. S. Poloczanska, A. J. Hobday, and A. J. Richardson, editors. A marine climate change impacts and adaptation report card for Australia 2009. NCCARF (National Climate Change Adaptation Research Facility), Publication 05/09, Queensland, Australia.

Hobday, A. J., A. D. M. Smith, I. C. Stobutzki, C.Bulman, R. Daley, J. M. Dambacher, R. A. Deng, J. Dowdney, M. Fuller, D. Furlani, S. P. Griffiths, D. Johnson, R. Kenyon, I. A. Knuckey, S. D. Ling, R. Pitcher, K. J. Sainsbury, M. Sporcic, T. Smith, C. Turnbull, T. I. Walker, S. E. Wayte, H. Webb, A. Williams, B. S. Wise, and S. Zhou. 2011. Ecological risk assessment for the effects of fishing. Fisheries Research 108:372–384.

Houde, E. D. 1987. Fish early life dynamics and recruitment variability. American Fisheries Society Symposium 2:17–29.

Houde, E. D. 1989. Subtleties and episodes in the early life of fishes. Journal of Fish Biology 35(Supplement A):29–38.

Houde, E. D. 2008. Emerging from Hjort's shadow. Journal of Northwest Atlantic Fishery Science 41:53–70.

Hubbert, M. K. 1969. Energy resources. Pages 157–242 *in* Resources and man: National Academy of Sciences National Research Council report of the committee on resources and man. Freeman, San Francisco.

Hutton, T., S. Mardle, and A. N. Tidd. 2008. The decline of the English and Welsh fishing fleet? Pages 26–48 *in* A. Payne, J. Cotter, and T. Potter, editors. Advances in fisheries science: 50 years on from Beverton and Holt. Wiley-Blackwell, New York.

Isaacson, W. 2007. Einstein: his life and universe. Simon and Schuster, New York.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293:629–637.

Jennings, S., and M. J. Kaiser. 1998. The effects of fishing on marine ecosystems. Advances in Marine Biology 34:201–352.

Jennings, S., and J. Rice. 2011. Towards an ecosystem approach to fisheries in Europe: a perspective on existing progress and future directions. Fish and Fisheries 12:125–137.

Jensen, O. P., T. A. Branch, and R. Hilborn. 2012. Marine fisheries as ecological experiments. Theoretical Ecology 5:3–22.

Jones, D. D., and C. J. Walters. 1976. Catastrophe theory and fisheries regulation. Journal of the Fisheries Research Board of Canada 33:2829–2833.

Kaplan, I. M., and B. J. McCay. 2004. Cooperative research, co-management and the social dimension of fisheries science and management. Marine Policy 28:257–258.

Kenny, A. J., H. R. Skjoldal, G. H. Engelhard, P. J. Kershaw, and J. B. Reid. 2009. An integrated approach for assessing the relative significance of human pressures and environmental forcing on the status of large marine ecosystems. Progress in Oceanography 81:132–148.

Laurel, B. J., and I. R. Bradbury. 2006. "Big" concerns with high latitude marine protected areas (MPAs): trends in connectivity and MPA size. Canadian Journal of Fisheries and Aquatic Sciences 63:2603–2607.

LCWCRTF (Louisiana Coastal Wetlands Conservation and Restoration Task Force). 1998. Coast 2050: toward a sustainable coastal Louisiana. Louisiana Department of Natural Resources, Baton Rouge.

Lindegren, M., C. Möllmann, A. Nielsen, and N. C. Stenseth. 2009. Preventing the collapse of the Baltic cod stock through an ecosystem-based management approach. Proceedings of the National Academy of Sciences of the USA 106:14722–14727.

Link, J. S. 2005. Translating ecosystem indicators into decision criteria. ICES Journal of Marine Science 62:569–576.

Link, J. S. 2010. Ecosystem-based fisheries management: confronting tradeoffs. Cambridge University Press, Cambridge, UK.

Litzow, M. A., and D. Urban. 2009. Fishing through (and up) Alaskan food webs. Canadian Journal of Fisheries and Aquatic Sciences 66:201–211.

Liu, J., T. Dietz, S. R. Carpenter, M. Alberti, C. Folke, E. Moran, A. N. Pell, P. Deadman, T. Kratz, J. Lubchenco, E. Ostrom, Z. Ouyang, W. Provencher, C. L. Redman, S. H. Schneider, and W. W. Taylor. 2007. Complexity of coupled human and natural systems. Science 317:1513–1516.

Lluch-Belda, D., D. B. Lluch-Cota, S. Hernandez-Vazquez, C. A. Salinas-Zavala, and R. A. Schwartzlose. 1991. Sardine and anchovy spawning as related to temperature and upwelling in the California current system. CalCOFI (California Cooperative Oceanic Fisheries Investigations) 32:105–111.

Lowe, M. E., and C. Carothers, editors. 2008. Enclosing the fisheries: people, places and power. American Fisheries Society, Symposium 68, Bethesda, Maryland.

Martell, S. J. D., T. E. Essington, B. Lessard, J. F. Kitchell, C. J. Walters, and C. H. Boggs. 2005. Interactions of productivity, predation risk, and fishing effort in the efficacy of marine protected areas for the central Pacific. Canadian Journal of Fisheries and Aquatic Sciences 62:1320–1336.

Morell, V. 2009. Can science keep Alaska's Bering Sea pollock fishery healthy? Science 326:1340–1341.

Murawski, S. A. 2007. Ten myths concerning ecosystem approaches to marine resource management. Marine Policy 31:681–690.

Murray, G., B. Neis, and D. C. Schneider. 2008. Lessons from a multi-scale historical reconstruction of Newfoundland and Labrador fisheries. Coastal Management 36:81–108.

Nehlsen, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. Fisheries 16(2):4–21.

NOAA (National Oceanic and Atmospheric Administration). 2009. NOAA coral reef conservation program: goals and objectives 2010–2015. NOAA Coral Reef Conservation Program, Silver Spring, Maryland.

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

NRC (National Research Council). 2002. Effects of trawling and dredging on seafloor habitat. National Academy Press, Washington, D.C.

NRC (National Research Council). 2006. Dynamic changes in marine ecosystems: fishing, food webs, and future options. National Academy Press, Washington, D.C.

Ommer, R. E., R. I. Perry, and B. Neis. 2008. Bridging the gap between social and natural fisheries science: why is this necessary and how can it be done? Pages 177–185 *in* J. L. Nielsen, J. J. Dodson, K. Friedland, T. R. Hamon, J. Musick, and E. Verspoor, editors. Reconciling fisheries with conservation: proceedings of the fourth world fisheries congress, volume 1. American Fisheries Society, Symposium 49, Bethesda, Maryland.

Pachauri, R. K., and A. Reisinger, editors. 2007. Contribution of working groups I, II and III to the fourth assessment report of the Intergovernmental Panel on Climate Change (IPCC). IPCC, Geneva, Switzerland.

Pauly, D. 2007. The *sea around us* project: documenting and communicating global fisheries impacts on marine ecosystems. Ambio 36:290–295.

Piet, G. J., H. M. Jansen, and M. J. Rochet. 2008. Evaluating potential indicators for an ecosystem approach to fishery management in European waters. ICES Journal of Marine Science 65:1449–1455.

Piet, G. J., R. van Hal, and S. P. R. Greenstreet. 2009. Modelling the direct impact of bottom trawling on the North Sea fish community to derive estimates of fishing mortality for non-target fish species. ICES Journal of Marine Science 66:1985–1998.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, J. Link, P. A. Livingston, M. Mangel, M. K. McAllister, J. Pope, and K. J. Sainsbury. 2004. Ecosystem-based fishery management. Science 305:346–347.

Pine, W. E., III, S. J. D. Martell, C. J. Walters, and J. F. Kitchell. 2009. Counterintuitive responses of fish populations to management actions: some common causes and implications for predictions based on ecosystem modeling. Fisheries 34:165–180.

Pitcher, T. J., D. Kalikoski, K. Short, D. Varkey, and G. Pramod. 2009. An evaluation of progress in implementing ecosystem-based management of fisheries in 33 countries. Marine Policy 33:223–232.

Plagányi, E. E. 2007. Models for an ecosystem approach to fisheries. FAO (Food and Agriculture Organization of the United Nations) Fisheries Technical Paper 477.

Pope, J. G., J. C. Rice, N. Daan, S. Jennings, and H. Gislason. 2006. Modelling an exploited marine fish community with 15 parameters: results from a simple size-based model. ICES Journal of Marine Science 63:1029–1044.

Rice, J. C. 2009. A generalization of the three-stage model for advice using the precautionary approach in fisheries, to apply broadly to ecosystem properties and pressures. ICES Journal of Marine Science 66:433–444.

Rice, J. C. 2011. Managing fisheries well: delivering the promises of an ecosystem approach. Fish and Fisheries 12:209–231.

Ring, I., B. Hansjürgens, T. Elmqvist, H. Wittmer, and P. Sukhdev. 2010. Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative. Current Opinion in Environmental Sustainability 2:15–26.

Robinson, L. A., S. Rogers, and C. L. J. Frid. 2008. A marine assessment and monitoring framework for application by UKMMAS and OSPAR: assessment of pressures of impacts—phase II application for regional assessments. University of Liverpool and Centre for the Environment, Fisheries and Aquaculture Science, contract C-08-0007-0027, Lowestoft, UK.

Rose, K. A., and J. H. Cowan Jr. 2003. Data, models, and decisions in U.S. marine fisheries management: lessons for ecologists. Annual Review of Ecology, Evolution, and Systematics 34:127–151.

Roy, C., P. Cury, A. Fontana, and H. Belvèze. 1989. [Spatio-temporal reproductive strategies of the clupeoids in West African upwelling areas]. Aquatic Living Resources 2:21–29. (In French.)

Sagan, C., and A. Druyan. 1992. Shadows of forgotten ancestors. Random House, New York.

Salas, F., H. Teixeira, C. Marcos, J. C. Marques, and A. Pérez-Ruzafa. 2008. Applicability of the trophic index TRIX in two transitional ecosystems: the Mar Menor lagoon (Spain) and the Mondego estuary (Portugal). ICES Journal of Marine Science 65:1442–1448.

Santos, A. M. P., M. D. F. Borges, and S. Groom. 2001. Sardine and horse mackerel recruitment and upwelling off Portugal. ICES Journal of Marine Science 58:589–596.

Seabrook, L., C. A. Mcalpine, and M. E. Bowen. 2011. Restore, repair or reinvent: options for sustainable landscapes in a changing climate. Landscape and Urban Planning 100:407–410.

St. John's Declaration. 2005. Declaration of ministers from the conference on the governance of high seas fisheries and the UN fish agreement: moving from words to action. Department of Fisheries and Oceans, St. John's, Newfoundland and Labrador, Canada. Available: www.dfo-mpo.gc.ca/fgc-cgp/declaration_e.htm. (March 2012).

Stige, L. C., G. Ottersen, K. Brander, K. S. Chan, and N. C. Stenseth. 2006. Cod and climate: effect of the North Atlantic oscillation on recruitment in the North Atlantic. Marine Ecology Progress Series 325:227–241.

Tietze, U., G. Groenewold, and A. Marcoux. 2000. Demographic change in coastal fishing communities and its implications for the coastal environment. FAO (Food and Agriculture Organization of the United Nations) Fisheries Technical Paper 403.

Travers, M., Y. J. Shin, S. Jennings, and P. Cury. 2007. Towards end-to-end models for investigating the effects of climate and fishing in marine ecosystems. Progress in Oceanography 75:751–770.

UN (United Nations). 1995. The United Nations agreement for the implementation of the provisions of the United Nations convention on the law of the sea of 10 December 1982 relating to the conservation and management of straddling fish stocks and highly migratory fish stocks (dated 8 September 1995). UN, Division for Ocean Affairs and the Law of the Sea, New York.

UN (United Nations). 2006. UN resolution 61/105: sustainable fisheries, including through the 1995 agreement for the implementation of the provisions of the United Nations convention on the law of the sea of 10 December 1982 relating to the conservation and management of straddling fish stocks and highly migratory fish stocks, and related instruments. UN, Division for Ocean Affairs and the Law of the Sea, New York. Available: www.un.org/Depts/los/general_assembly/general_assembly_resolutions.htm. (March 2012).

UN (United Nations). 2009. UN resolution 64/72: sustainable fisheries, including through the 1995 agreement for the implementation of the provisions of the United Nations convention on the law of the sea of 10 December 1982 relating to the conservation and management of straddling fish stocks and highly migratory fish stocks, and related instruments. UN, Division for Ocean Affairs and the Law of the Sea, New York. Available: www.un.org/Depts/los/general_assembly/general_assembly_resolutions.htm. (March 2012).

UNEP (United Nations Environment Programme). 2012. UNEP 2011 annual report. UNEP, Division of Communications and Public Information, Nairobi, Kenya.

Walters, C. J. 1986. Adaptive management of renewable resources. Macmillan, New York.

Walters, C. J. 2007. Is adaptive management helping to solve fisheries problems? Ambio 36:304–307.

WCED (World Commission on Environment and Development). 1987. Our common future. Oxford University Press, Oxford, UK.

Wilson, E. O. 1998. Consilience: the unity of knowledge. Knopf, New York.

Worm, B., E. B. Barbier, N. Beaumont, J. E. Duffy, C. Folke, B. S. Halpern, J. B. C. Jackson, H. K. Lotze, F. Micheli, S. R. Palumbi, E. Sala, K. A. Selkoe, J. J. Stachowicz, and R. Watson. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science 314:787–790.

Worm, B., R. Hilborn, J. K. Baum, T. A. Branch, J. S. Collie, C. Costello, M. J. Fogarty, E. A. Fulton, J. A. Hutchings, S. Jennings, O. P. Jensen, H. K. Lotze, P. M. Mace, T. R. McClanahan, C. Minto, S. R. Palumbi, A. M. Parma, D. Ricard, A. A. Rosenberg, R. Watson, and D. Zeller. 2009. Rebuilding global fisheries. Science 325:578–585.