NRC Research Press

PERSPECTIVE

The art of ecosystem-based fishery management

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Abstract: The perception that ecosystem-based fishery management is too complex and poorly defined remains a primary impediment to its broadscale adoption and implementation. Here, I attempt to offer potential solutions to these concerns. Specifically, I focus on pathways that can contribute to overall simplification by moving toward integrated place-based management plans and away from large numbers of species-based plans; by using multispecies or ecosystem models and indicators that permit the simultaneous and consistent assessment of ecosystem components while also incorporating broader environmental factors; and by consolidating individual administrative and regulatory functions now mostly dealt with on a species-by-species basis into a more integrated framework for system-wide decision-making. The approach focuses on emergent properties at the community and ecosystem levels and seeks to identify simpler modeling and analysis tools for evaluation. Adoption of ecosystem-based management procedures relying on simple decision rules and metrics is advocated. It is recommended that we replace static concepts for individual species focusing on maximum sustainable yield with a dynamic ecosystem yield framework that involves setting system-wide reference points along with constraints to protect individual species, habitats, and nontarget organisms in a dynamic environmental setting.

Résumé : La perception voulant que la gestion écosystémique des pêches soit trop complexe et mal définie demeure un des principaux obstacles à son adoption et son application à grande échelle. Je tente donc d'offrir des pistes de solution à ces préoccupations. J'aborde plus particulièrement des avenues qui pourraient contribuer à simplifier globalement cette approche en l'orientant sur des plans de gestion intégrés axés sur l'emplacement plutôt que sur un grand nombre de plans axés sur des espèces données; en utilisant des modèles et indicateurs multi-espèces ou écosystémiques qui permettent l'évaluation simultanée et cohérente de différents éléments de l'écosystème tout en intégrant des facteurs environnementaux plus larges; et en consolidant les différentes fonctions administratives et de réglementation qui, à l'heure actuelle, font principalement l'objet d'une approche espèce-par-espèce, en un cadre décisionnel plus intégré à portée systémique. L'approche met l'accent sur les propriétés émergentes à l'échelle de la communauté et de l'écosystème et cherche à cerner des outils de modélisation et d'analyse simplifiés pour les fins d'évaluation. L'adoption de procédures de gestion écosystémique reposant sur des règles de décision et des paramètres simples est préconisée. Il est recommandé de remplacer les concepts statiques visant des espèces individuelles et axés sur le rendement équilibré maximum par un cadre de rendement écosystémique dynamique qui comprend l'établissement de points de référence d'échelle systémique et de contraintes visant la protection des différentes espèces, des habitats et des organismes non ciblés dans un contexte environnemental dynamique. [Traduit par la Rédaction]

Introduction

Despite long-standing calls for incorporation of broader ecological principles in fisheries management, implementation on a global scale remains slow and tenuous (Pitcher et al. 2009). The scientific foundations for ecosystem-based fishery management (EBFM) have been established over the last several decades (see, for example, Watt 1968; Wagner 1969; Cushing 1975; Regier 1978; Stroud and Clepper 1979; Mercer 1982; Pitcher and Hart 1982; May 1984; Caddy and Sharp 1986; Daan and Sissenwine 1990; Mooney 1998; AKSGP 1999; Hall 1999; Jennings et al. 2001; Sinclair and Valdimarsson 2003; Walters and Martell 2004; Browman and Stergiou 2004; and contributions therein). Recent books, symposia, and dedicated journal volumes reveal a very active and productive field of inquiry (Fowler 2009; Link 2010; Christensen and McLean 2011; Belgrano and Fowler 2011; Glazier 2011; Essington and Punt 2011; Fanning et al. 2011; Stephenson et al. 2012; Bundy et al. 2012; and Kruse et al. 2012). These advances notwithstanding, important concerns have been raised related to the overall tractability, cost, and potential effectiveness of incorporating ecosystem considerations in tactical fisheries management strategies (e.g., Longhurst 2006, 2010; Hilborn 2011; Rice 2012; Cowan et al. 2012). In the following, I attempt to provide some possible pathways toward resolution of these concerns.

EBFM is intended to provide an integrated framework for the sustainable delivery of a key ecosystem service. It takes into account interrelationships among the elements of the system, considers humans as an integral part of the ecosystem, and accounts for environmental influences. As defined here, EBFM is a place-based rather than a species-based approach. EBFM is designed to be adaptive in response to changing conditions and as scientific understanding accrues. It accounts for uncertainty and the mix of different (and potentially competing) societal goals and objectives. EBFM differs from what is sometimes referred to as an ecosystem approach to fishery management (EAFM), which retains a primary focus on individual species, stocks, or fisheries while incorporating ecosystem considerations into the whole.[1] In con-

Received 5 April 2013. Accepted 13 November 2013.

Paper handled by Associate Editor Kenneth Rose.

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[1]The proliferation of terms related to the general concept of holistic approaches to fishery management often masks a basic commonality of concepts and intent (Arkema et al. 2006). Here I simply wish to distinguish between approaches that focus on integrated management plans for defined ecological regions and those in which species and their associated fisheries remain the focal points for management in the context of broader ecological considerations.

 Published at www.nrcresearchpress.com/cjfas on 28 November 2013.

trast, the spatial orientation of EBFM fits more naturally within the broader domain of ecosystem-based management (EBM) (Fogarty and McCarthy 2014). EBM addresses the cumulative impacts of the broad spectrum of human activities affecting ecosystems. The clear direction of national and international initiatives now underway is toward EBM and away from a sole focus on individual issues such as fisheries, coastal development, water quality, etc. In this context, EBFM is just one element of EBM; within this framework, objectives for EBFM must be reconciled with those of other sectors.

An overarching goal of EBM is to protect ecosystem structure and function to ensure the continued flow of ecosystem services. This utilitarian framework is not intended to downplay the intrinsic importance of these systems but rather to focus attention on our responsibility to actively manage the spectrum of human activities affecting aquatic ecosystems. Preservation of diversity in biological, social, and economic subsystems emerges as a critical element in meeting this goal. Palumbi et al. (2009) propose that preservation of biodiversity can serve as a cornerstone for EBM. Parallel considerations for the human dimension of EBM and EBFM are no less important. Management strategies that constrain options of fishers to adapt to changing conditions can lead to unintended consequences and increased stress on aquatic ecosystems. For social-ecological systems, maintaining diversity at all levels provides a buffer against uncertainty and a hedge against future change.

The point of departure for this essay is not that conventional fisheries management has universally failed but rather that it is necessarily incomplete. It can only take us so far. When clearly defined targets and limits for management have been established and enforced, it has stemmed the tide of overexploitation in a number of fishery ecosystems around the world (e.g., Mace 2001; Worm et al. 2009; Worm and Branch 2012). Single-species approaches, however, ultimately do not lead to an internally consistent framework for management of assemblages of interacting species. Further, they do not generally account for changing environmental conditions or for the broader human dimensions of fishery systems (Charles 2001; Garcia and Charles 2008). For reasons described below, as fishing mortality rates are brought under control, EBFM becomes more rather than less important. Conventional management approaches unavoidably set up conflicts among individual management plans by ignoring interactions among species and the trade-offs that inevitably emerge. Here it is argued that we must build on the hard-won insights and successes of conventional assessment and management approaches and take the next steps toward a more holistic ecosystem framework to address these issues.

Fishery *science* is often described as principally focusing on individual species and their dynamics. A brief tour through the literature in fisheries journals, some in continuous publication for over a century, should be sufficient to quickly dispel this view. The origins of fishery science rest in a multidisciplinary framework as reflected in the founding principles of a number of aquatic research and management institutions established in the 19th century (Smith 1994). It is unquestionably true that fisheries "management" has largely centered on individual species and stocks. In this, fisheries management shares a connection with other areas of applied ecology such as conservation biology in which modeling efforts in support of management have often concentrated on individual species of concern. The rich tradition of broader-based multidisciplinary research in the ecology of exploited aquatic systems, however, does provide a strong foundation for addressing the scientific requirements for EBFM. In a real sense, EBFM entails coming full circle to the roots of the discipline.

Viewed in the proper light, adoption of EBFM offers avenues to simplification of current management approaches.[2] The broader EBFM perspective affords opportunities for consolidating assessments and management plans for a very large number of individual species or stocks into a more cohesive and integrated set for defined ecological regions. Successful implementation of EBFM will ultimately depend on finding ways of managing scientific, administrative, and regulatory complexity. It will require skill in the arts of effective communication, stakeholder engagement, and simplification in the face of apparent complexity. The art of negotiation will be no less essential as trade-offs are identified and resolution is sought.

Background

The fundamental limitations of the prevailing single-species approach and associated management reference points have long been appreciated. Interspecific interactions, environmental and climate influences on system-wide productivity, and other factors all have a direct effect on the appropriate choice of limits and targets for management. These considerations call for a dynamic rather than static concept of management reference points. Three early perspectives will suffice to highlight the recognized limitation of single-species maximum sustainable yield (MSY) as a management objective:

> … it is very doubtful if the attainment of maximum sustainable yield from any one stock of fish should be the objective of management except in exceptional circumstances. (Gulland 1969)
>
> … it seems improbable that the perfect strategy would be to take MSY from each species. (Larkin 1977)
>
> … common sense should lead us to dismiss a concept of optimum yield drawn from a series of single-species MSYs. (Sissenwine 1978)

Ignoring interspecific interactions is identified by each of these authors as a central limitation of the single-species approach. An important corollary is that natural mortality is not constant with age or size, nor is it time-invariant. Yet analyses embodying these assumptions remain prevalent. Incorrectly specifying a constant natural mortality rate in a single-species system introduces a scaling error that can be largely offset in the specification of management reference points. However, in a multispecies context as assemblages of interacting species change in response to management actions and (or) natural fluctuations, resulting in time-varying natural mortality rates, a much more insidious problem is introduced.

Single-species MSY continues to be a cornerstone of current management practices in many parts of the world. The early concentration on single-species management models no doubt arose from legitimate concerns related to analytical and regulatory tractability. Adoption of the MSY concept in national and international conventions also appears to have been strongly driven by geopolitical imperatives (Finley 2008, 2010). As noted by Mace (2001), the switch to considering MSY as a limit rather than a target reference point has played an invaluable role in reducing overexploitation in many areas. An unquestionable merit of MSY-related reference points has been the adoption of clearly defined standards for assessment and management. In the United States, under the provisions of the Magnuson–Stevens Fishery Conserva-

[2]This is not meant to imply that it is a simple problem. It is in fact a "wicked" problem (Berkes 2012) in which predictability is limited and unanticipated change is likely. It is nonetheless necessary to find pathways toward simplification if EBFM is to be tenable. The problems identified here do not go away if ignored. If not directly confronted, they will lead to unintended consequences.

tion and Management Act, optimum yield is defined as maximum sustainable yield as **reduced** by relevant social, economic, and ecological considerations. However, it remains relatively uncommon for reference points to be adjusted in this way based on ecological considerations. While the early concerns cited above refer specifically to MSY-related metrics, they are relevant to many of the MSY-proxy reference points now commonly in use that also ignore interspecific interactions, environmental variability, and other ecological considerations.

When ecosystems have been degraded by intensive fishing, resulting in stock collapses and alterations in ecosystem structure and function, the first steps for remedial action are effectively identical under both single-species and ecosystem approaches to management: sharply reduce fishing pressure (Mace 2001). This has led many commentators to note with justification that effective single-species management goes a long way toward meeting the needs of EBFM. But the issue is ultimately deeper and more systemic than controlling fishing pressure on individual species viewed in isolation. As fishing mortality rates are brought under control, interspecific interactions, climate and environmental forcing, and other factors become more important relative to the effects of fishing and therefore more critical to address. They are no longer masked by the overriding effects of overexploitation. If biological interactions are important, then trying to optimize the yield from individual species without accounting for these effects can only result in misleading management advice and expectations. When interacting species are covered by separate management plans, these plans unavoidably and actively work at cross-purposes in their attempts to achieve biomass levels corresponding to (single-species) MSY or to meet related objectives.

Results from a wide spectrum of multispecies and ecosystem models support the view that simultaneously extracting single-species MSYs from an assemblage of interacting species is not possible (e.g., Brown et al. 1976; Collie and Gislason 2001; Mueter and Megrey 2006; Steele et al. 2011; Walters et al. 2005; Mackinson et al. 2009; Fogarty et al. 2012; Heath 2012). The problem is etched in sharp relief when considering mixed-species fisheries where species-specific catchability and vulnerability to fishing result in different outcomes for each under a common level of fishing effort. We cannot fully control the fishing mortality rates separately for the individual species composing the multispecies assemblage. Differential mortality rates for different parts of the system will in turn lead to changes in community structure. Adopting the EBFM perspective does not obviate this problem; it does ensure, however, that it will be dealt with in a transparent way and not ignored. The centrality of the mixed-species problem was recognized over 50 years ago by McHugh (1959), who called for "management en masse" — a perspective that anticipated the use of aggregate production models described below (see also McHugh 1988).

If EBFM is to successfully replace current single-species approaches, unambiguous reference points and standards at the community and ecosystem levels must be established. It has long been recognized that, within limits, total fish yield, size structure, and biomass levels often reflect remarkably conservative properties of aquatic ecosystems and communities (Kerr and Ryder 1988). The apparent greater stability at the system level may reflect overall energetic constraints on system dynamics. We can take advantage of these properties to establish system-wide protocols for EBFM to ensure that system resilience can be maintained.

Coping with complexity

Whether we can deal with the daunting complexity of ecosystems and the associated management challenges is indeed a legitimate concern. We need to recognize limits to our understanding, precision, and control in the assessment and management of fishery systems. Substantial increases in administrative and regulatory efficiency are possible by replacing large numbers of management plans for individual species or stocks with a much smaller number of fully integrated place-based plans. Here, a focus on system-wide production potential is advocated. The productivity of any ecosystem is ultimately set by the amount of energy fixed at the base of the food web, placing constraints on the production of all species, including ones of economic importance. This production is further conditioned on changing environmental states and must be viewed in a dynamic context. By shifting from a single-species to a community or ecosystem perspective but developing production-based ecosystem reference points, a natural bridge to current management practices can be established.

Scientific complexity

Models in support of EBFM can be arrayed along a continuum of complexity involving trade-offs in realism, mechanistic detail, and parameter and (or) model uncertainty. A central lesson in forecasting drawn from a diverse set of fields is that bigger, more complex models are not necessarily better and that model overfitting is a pervasive and pernicious problem (Silver 2012; Pilkey and Pilkey-Jarvis 2007). Gunderson and Holling (2002) indicate that a model should have no more than a handful of variables if it is to remain tractable and understandable. Single-species assessment models and approaches have arguably grown too complex with respect to data availability and quality, transparency to stakeholders, and other concerns (Cotter et al. 2004). For obvious reasons, these problems can be considerably amplified under EBFM unless a strategy for deliberately coping with complexity is adopted (Hill et al. 2007).

Models of low to intermediate complexity can often outperform more complicated models in forecast skill (e.g., Silvert 1981; Ludwig and Walters 1985; Walters 1986; Costanza and Sklar 1985; Fulton et al. 2003; Grimm et al. 2005; Hannah et al. 2010; Plagányi et al. 2012). This general point has been framed in different but interrelated ways including the trade-off between systematic bias and measurement error (Walters 1986), interconnectedness (Costanza and Sklar 1985), and "payoff" (Grimm et al. 2005), all as a function of model complexity. Grimm et al. (2005) adapted the concept of the Medawar zone to describe the region of optimal payoff at intermediate levels of model complexity. This designation honors Sir Peter Medawar, who memorably described science as the "art of the soluble" (Medawar 1967). In this context, payoff refers to levels of model complexity that provide higher levels of predictability (Grimm et al. 2005).

Models for EBFM

The approach taken to assessing the status of communities and ecosystems for EBFM will ultimately depend on the choice of management objectives and the nature of the scientific information and infrastructure available in different areas. These elements will differ substantially in different parts of the world. A range of methods and approaches that can span a broad spectrum of needs and available resources is therefore required.

The appropriate choice of modeling approaches depends critically on these issues and the specific requirements the model is intended to meet (Silvert 1981). The models described below span a range of complexities that can be tailored to the needs and scientific resources available in different areas. Because models at the more complex end of the spectrum have been nicely covered in recent reviews (e.g., Plagányi 2007), here I will focus on simpler models with modest data requirements that may be broadly applicable in regions where data availability and scientific resources are more constraining.

Even in extremely data-limited situations, it is nevertheless possible to make first-order estimates of expected system-level yield using broadly available data. The simplest approaches to estimating potential fish yields are based on empirical models. Predictive models relating total yields to chlorophyll concentration and (or) primary production have been applied in both marine (e.g., Ware and Thomson 2005; Frank et al. 2006; Chassot et al. 2010; Friedland et al. 2012) and freshwater systems (e.g., McConnell et al. 1977). When applied on a regional basis, these predictors reveal strong evidence for bottom-up controls on fish yields in many ecosystems. These empirical statistical descriptors are very much in the spirit of macro-ecological approaches (Brown 1995; Maurer 1999) designed to complement experimental approaches and other modeling perspectives. Simple food chain models have also been used to assess fishery production potential in marine ecosystems (Pauly 1996; Ware 2000). Extensions to this approach using new information on energetic pathways and refinement of key inputs such as ecological transfer efficiencies have been developed (Fogarty et al. in press). In freshwater systems, a strong tradition of empirical yield models incorporating geomorphological characteristics, nutrients, and other factors has been established (Ryder 1965; Kerr and Ryder 1988).

Predictive models capitalizing on new developments in nonlinear time series analysis are also now being applied to catch and abundance series to characterize system dynamics and to develop short-term forecasts (Glaser et al. 2013). They build on the crucial insight that for systems exhibiting nonlinear dynamics, information on the system as a whole is encoded in time series for one or more individual parts. For systems with an important deterministic component, this broader system information can, in principle, be recovered by reconstructing the underlying attractor in a time-delayed coordinate system (Takens 1981; Deyle and Sugihara 2011). The method uses out-of-sample forecast skill as the measure of model performance and is consonant with earlier calls for the development of a predictive science of ecology (Peters 1991). Nonlinear time series analysis has been used to assess co-predictability in multispecies systems, where a model developed for one species of a potentially interacting pair is applied to the other and forecast skill is assessed (Liu et al. 2012). Full multivariate nonlinear times series methods afford opportunities to examine causal linkages among ecosystem components (Sugihara et al. 2012) including the effects of fishing and climate forcing on system dynamics (Deyle et al. 2013). These nonparametric models offer an alternative approach to dealing with model uncertainty.

In data-rich regions of the globe a broader range of options for analysis is possible, including application of multispecies biomass dynamics models, biomass- and size-spectrum models, size- and age-structured multispecies models, and full ecosystem models. Multispecies production models in which pairwise interactions between species are specified have been applied in both freshwater and marine systems (Walter and Hoagman 1971; Pope 1976; May et al. 1979; Sissenwine et al. 1982; Sullivan 1991). This approach is likely to be most tractable in systems with relatively few species. For example, Sullivan (1991) applied this method to a three-species Baltic Sea fish community and found evidence for both direct and indirect species interactions. In models of this type, the magnitude and sign of empirically determined interaction terms are used to assess the type of interaction involved (competition, predation, etc.). In systems of higher dimensionality, the data requirements with respect to time series length become more constraining for our ability to detect interactions (Sissenwine et al. 1982).

With appropriate care, the complexity of the system can be reduced by applying different aggregation strategies. Hilborn and Walters (1992, p. 449) noted that a "lump the species together" approach offered perhaps the best prospects for success for multispecies assessment and management among the alternatives they considered. It has the twin virtues of simplicity and broad applicability to fishery systems throughout the world because of its modest data requirements. Although the method has been employed to remedy data limitations and (or) address system complexity (Sugihara 1984), the potential to implicitly account for interspecific interactions has also been a motivating factor in its use (Brown et al. 1976). In this approach, the trajectory of the whole is taken to integrate the effects of fishing and species interactions on the parts. For recent examples, see the contributions in Bundy et al. (2012).

It is essential to recognize that development of models for aggregate-species groups is not directed at understanding the dynamics of the individual species in the assemblage. Rather, we are seeking to base our assessment on the properties of the assemblage as a whole. These properties cannot be reconstructed by studying the parts in isolation. For nonlinear systems, the properties of the whole are not the same as those of its parts. In particular, understanding emergent properties (von Bertalanffy 1968) is a critical consideration. I believe that the primary rationale for focusing on functional groups is not mere convenience; rather, they are key structural elements in the way that the system operates. It must also be stressed that if we use aggregate models to set reference points, it will also be necessary to continue to track individual species (where feasible) and to set precautionary buffers to protect vulnerable species within aggregate groups (e.g., Fogarty et al. 2012; Gaichas et al. 2012; Nesslage and Wilberg 2012).[3] See Tyler et al. (1982) for a related discussion of management of assemblage production units.

Aggregate-species production models have been applied to entire fishery ecosystems (e.g., Brown et al. 1976; FAO 1977; Mueter and Megrey 2006), individual functional groups (Sparholt and Cook 2010; Fogarty et al. 2012), and a collection of functional groups with explicit interaction terms connecting the groups (Ralston and Polovina 1982; Bell et al. in press). These simple multispecies biomass dynamics models can readily accommodate environmental covariates to account for changing physical or ecological conditions (Mueter and Megrey 2006; Fogarty et al. 2012). More complex models for guilds or functional groups that incorporate broader demographic or ecological features have also been developed (Collie and DeLong 1999; Steele et al. 2011; Heath 2012).

Membership rules for defining functional groups are critically important. It is recommended here that functional groups should comprise species that are caught together, have similar life history characteristics, and occupy similar trophic positions. Such groups will, inter alia, often share similar size characteristics, habitat preferences, and history of anthropogenic and environmental perturbation. This definition therefore extends the guild concept in fisheries management (e.g., Austen et al. 1994) to accommodate a broader set of fishery-related and scientific considerations. A functional group can be thought of as a portfolio of species sharing certain common characteristics (see Hanna 1998; Edwards et al. 2004; and Sanchirico et al. 2008 for more on the portfolio concept in a multispecies fishery context). As with financial instruments, constructing a portfolio containing elements with negative covariances among at least some components can provide an important hedge against uncertainty and risk.

Methods of aggregation based on size or biomass structure have also been successfully used to represent multispecies systems (see Kerr and Dickie 2001). The approach takes advantage of the conservative properties of demographic structure in fishery systems (e.g., Murawski and Idoine 1992). Pope et al. (2006) and Jennings et al. (2008) provide recent size-based analyses that show considerable promise in capturing key ecosystem characteristics and

[3]This does not imply that full single-species analyses are required. Metrics that have been aggregated to represent system properties should also always be carefully examined in their disaggregated form to track the status of the component parts and to identify the need for corrective measures to protect individual parts of the system where necessary.

dynamics while focusing on a restricted number of parameters to define the system.

At the next level of complexity, multispecies and ecosystem models tracking individual species have been developed for fishery systems (see reviews in Hollowed et al. 2000; Whipple et al. 2000; Plagányi 2007). A substantial global initiative in applying the EcoPath with EcoSim (EwE) modeling framework has been developed and is being used to support EBFM in many parts of the world. Christensen et al. (2009) provide initial results for a prototype EwE "database-driven" system for each of the 66 currently designated Large Marine Ecosystems around the world. The analysis draws on a set of global databases to provide an initial parameterization of EwE models that can then be subsequently refined by local experts. End-to-end models such as Atlantis (Fulton et al. 2011) have also been developed for more than 30 systems around the world (B. Fulton, CSIRO, personal communication).

The simpler models described above are principally "top-down" (Silvert 1981) approaches that focus on higher levels of ecological organization. Depending on their internal structure, they may or may not be able to represent complex dynamical behaviors. Agent-based models provide an alternative "bottom-up" approach. These methods can be computationally intensive but employ simple decision rules for individual elements of the system (e.g., Grimm et al. 2005; Railsback and Grimm 2012). These simple rules can in some cases generate quite complex dynamics (e.g., regime shifts) at the system level. In fisheries ecology, their use has most often been in the form of individual-based models. Grimm et al. (2005) note that direct consideration of observed patterns in the dynamics of these systems can substantially aid in guiding and constraining complexity in agent-based models. Here, we would focus on properties such as stationarity, variability, and resilience for the whole and the parts. Increasing interest in the potential utility of agent-based and multiagent models by social scientists (e.g., Gilbert 2008) may provide one avenue for a fuller integration of the social and natural sciences for EBFM and EBM. To date, ways of quantitatively connecting broad social and ecological considerations in EBFM have been limited (Garcia and Charles 2008; but see Hennessey and Sutinen 2005 and Holland et al. 2010).

Indicators

Indicators are central components of the methods and modeling approaches for EBFM. They serve as key elements of Integrated Ecosystem Assessments (IEAs; Levin et al. 2008, 2009, 2013) and ecosystem-based management procedures (EBMPs; Sainsbury et al. 2000). Guidelines for selection of informative indicators have been set forth by a number of authors (e.g., Jennings 2005; Rice and Rochet 2005; Link 2010). Here I will focus on the role of indicators to supplement some of the modeling methods described above and as elements of IEAs and EBMPs. Models that do not explicitly include consideration of demographic structure, spatial dynamics, environmental drivers (or other externalities), and social drivers can be complemented by consideration of available indicators that reflect these dimensions. Metrics that can provide leading indicators of rapid shifts in state (e.g., increases in variance and (or) autocorrelation) can be an important adjunct to models that cannot otherwise represent complex dynamical behavior.[4] Finally, given the importance ascribed earlier to maintaining diversity in biological, social, and economic subsystems, indicators that track changes in diversity of these components can be invaluable.

A list of candidate indicator categories to meet these requirements might include:

- Key environmental and climate indicators for oceanographic and (or) atmospheric conditions
- Catch and landings by species and (or) functional groups and fishing effort (where available)
- Biomass, abundance, or production by species and (or) functional groups at a number of trophic levels from plankton to apex predators
- Species diversity of biological communities and catches and diversity of fishing fleet characteristics
- Diversity in size and (or) age composition size or biomass spectra of biological communities and in catch or landings
- Spatial concentration indices for biological communities and for fishing fleets
- Ecosystem-balance indicators (e.g., the ratio of piscivores to planktivores)
- Mean trophic level in the ecosystem and in the catch
- Levels of employment, net revenues, and (where possible) profits
- Measures of social well-being in fishing communities
- Change in variance and (or) autocorrelation in space and time for any of these indicators

Although availability of this entire suite of indicator categories will vary widely in different settings around the world, elements of the uppermost tier in this list should be broadly accessible. When both ecosystem pressure and state variables are available, it may be possible to directly establish reference points and control rules for EBFM within an indicator framework. Samhouri et al. (2012) provide examples of how this might be accomplished for indicators encompassing a range of levels of complexity and ability to represent ecosystem pressures and states (see also Large et al. 2013). Qualitative depiction of indicators in the form of traffic light-style representations can be readily adapted for use in EBFM (Caddy 2002), and decision rules can be devised and implemented using fuzzy control systems or other methods. If analysis of a set of indicator variables indicates vulnerabilities not detected or represented in multispecies assessment models, appropriate precautionary measures should be adopted.

Administrative and regulatory complexity

The current structure supporting the machinery of single-species stock assessment and management in the developed world is an immensely complex enterprise. National fisheries agencies and international bodies support a very large number of working groups, each involving substantial representation and charged with developing stock assessments for individual species. The assessment process further entails a formal peer review process for each of the individual assessments. Collectively, these assessment and review elements incur very significant administrative costs in the developed world.

A full EBFM approach would consolidate the number of required working groups and modeling structures into a much more tractable number charged with developing integrated assessments for defined ecoregions (see below). Group membership, representing a wider array of disciplines ranging from climatology, physical science, and fisheries ecology to social science, would be larger and much more diverse than a typical individual species or stock working group. It is very likely that increased diversity in scientific and stakeholder representation in working groups will present new challenges in reaching consensus and expert facilitators will be essential in finding common ground. It must be anticipated that this process initially will be very time consuming as protocols are developed and agreement is sought.

A focus on regulatory complexity will be particularly important in EBFM. Conventional single-species approaches with strong top-down controls have inexorably led to increasing complexity in management, often with adverse outcomes (e.g., Healey and Hennessey 1998; Cochrane 1999). The pursuit of perceived levels of fairness in allocation procedures ultimately breaks down un-

[4]For potential limitations related to time series length and precision, see Perretti and Munch (2012).

der the demands placed on data and regulatory control (Healey and Hennessey (1998). Rigid command and control systems lead to brittle management structures that are prone to failure (Gunderson and Holling 2002). Simple decision rules for EBFM coupled with adaptive management structures informed by strong input from stakeholder groups representing a range of interests will again be essential. The temptation to add complexity should be resisted and only adopted after costs and benefits are carefully evaluated (Cochrane 1999).

Cost considerations

In addition to cost considerations related to the administrative and regulatory systems, the overall issue of cost of ecosystem monitoring is an important concern. For the simplest of the modeling approaches described above, requisite satellite-derived information on chlorophyll concentration is broadly available, as is catch or landings data (although data quality may not be consistent among areas). It is worth noting that the Global Environment Facility is now investing heavily in capacity building and providing the resources needed to guide sustainable development and management of fishery systems in the developing world under the aegis of the Large Marine Ecosystem concept (Sherman 2005; see also http://www.thegef.org/gef/news/recovering-ocean-health).

In many parts of the developed world, fisheries agencies and other institutions have implemented far-reaching observing programs that encompass some or all of the following: operational physical oceanography, plankton dynamics, trophic interactions, habitat, protected and nontarget species monitoring, and other ecosystem elements. The long-standing recognition that we require broader-based ecological understanding for effective fisheries management guided the establishment of these programs, providing a very rich source of ecological information to inform EBFM and its elements (IEAs, EBMPs, etc.). Fishery-independent trawl surveys are underway in many parts of the world and are now being used in single-species and multispecies models (for a compilation, see Ricard et al. 2012; http://ramlegacy.marinebiodiversity.ca/). Other invaluable long-standing ecosystem monitoring programs include the Continuous Plankton Recorder surveys operated by the Sir Alister Hardy Foundation for Ocean Science (SAHFOS). Collectively, these trawl and plankton surveys go far beyond immediate needs for single-species assessment models and have been used in the development of ecosystem indicators and multispecies or ecosystem models. Further, the information collected in these fishery-observing programs is now routinely being used to provide important insights into a much broader array of issues, including assessing ecosystem changes related to climate variability.

When this information is not used in the development of management advice, we are not capitalizing fully on our research investments. In these instances, the problem is not that we cannot afford to collect the information needed for EBFM, but rather that we are not effectively using it. It is of course possible that the cost of these programs cannot be borne indefinitely. In this case, it will be necessary to identify the programs that provide the most informative data and match them to management objectives in different regions to assign priorities.

Ecosystem-based management procedures

The issues of cost and complexity in conventional fishery assessment and management have been motivating factors in developing a simpler management procedure approach (Butterworth et al. 1997; Butterworth 2007; Rademeyer et al. 2007).[5] Management procedures (MPs) entail the specification of a potentially simple set of rules for translating information from an assessment model into a management action. There is binding agreement beforehand on factors such as the model choice, associated data, and the actions to be taken if a management threshold is crossed. Ways of adaptively coping with unanticipated change can be built into the procedure. MPs can remain in place for multiyear (3–5 years) time frames and can be explicitly structured to enhance prospects for stability in the fishery by modulating the amount of change from one time step to the next, providing a more manageable time horizon for business, scientific, and administrative planning. The performance of alternative MPs is rigorously evaluated by simulation with respect to factors such as yield and (or) profitability, uncertainty, and risk before any actual implementation is considered. A key question is, Can simpler approaches provide a workable solution with acceptable performance characteristics?

There is a compelling connection between this question and an approach advanced by Herbert Simon, a Nobel Laureate in Economic Sciences, who coined the term “satisficing” (Simon 1956, 1996).[6] Simon questioned whether a complex optimization framework is in fact preferable to simpler heuristic methods when the full costs involved with the former are considered. Simon argued that we often cannot fully evaluate all alternatives and that we frequently do not have all the necessary information to make “optimal” decisions — a recognition of the need for a system based on “bounded rationality”. A satisficing solution is one that yields a defensible outcome that meets defined objectives and is satisfactory to the end users. It resonates with Alec MacCall’s concept of “Pretty Good Yield” (Hilborn 2010), in which analytical limitations in defining optima are clearly recognized. It is entirely consistent with the viewpoint adopted in developing management procedures, where pragmatism is a critically important consideration. Given the complexity of ecosystem dynamics, uncertainties in our understanding, and the interwoven strands of a diverse set of human activities affecting aquatic ecosystems, it may in fact be appropriate to acknowledge that we are, at best, in a position to offer satisficing solutions. One could argue that current management, while considering results based on optimization procedures in stock assessment, often defaults to a satisficing solution when integrating broader social and economic considerations with conservation needs. Satisficing solutions involve an evaluation of past experience and application of rules of thumb, although more sophisticated approaches involving game theory (Sterling 2003), fuzzy logic (Goodrich et al. 1999), and agent-based models (Railsback and Grimm 2012) can be applied. When we are dealing with trade-offs involving incommensurable objectives, a satisficing approach might be the most fruitful avenue to pursue.

One of the earliest and perhaps best-known EBMPs was developed for krill (*Euphausia superba*) in the Southern Ocean by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) (De la Mare 1996; Constable 2002). The overarching goal of CCAMLR is to maintain ecological relationships among harvested, dependent, and related species and to restore depleted populations within the convention area. Krill occupy the nexus of the Southern Ocean food web and many fish, mammal, and bird species are dependent on krill as prey. The objective for the krill management procedure is to maintain spawning stock biomass (SSB) at three quarters of the unexploited level to ensure adequate food supplies for predators. A stochastic population model serves as the operating model. The decision rule for selecting a target exploitation rate involves consideration of the probability that the median SSB escapement level will be 75% over a 20-year period and the probability that SSB will be driven below

[5]For caveats see Rochet and Rice (2009).

[6]For an early discussion of satisficing in a fishery context, see Opaluch and Bockstael (1984).

20% of the target escapement level at the end of the 20-year period with no more than a 10% probability (Constable 2002). The lower of the two exploitation rates meeting these requirements is selected for implementation. Other objectives and decision rules explicitly considering krill-dependent species have also been proposed (Constable 2002).

Elements of a prototype EBMP

Although many potential paths for specifying EBMPs can be identified, here I will focus on one example that I believe may be broadly applicable and can accommodate the broad spectrum of information and scientific resources in different parts of the globe. The approach centers on a hierarchical specification of system-wide limit reference points to set overall harvest constraints and then establishing constraints on catch levels and practices to protect individual ecosystem components. Additional ecological, social, and economic considerations will also be factored into the constraints at this latter stage.

The main elements are:

- Select spatial management units
- Establish specific management objectives, reference points, and decision rules
- Agree on tactical modeling approaches and associated data to assess ecosystem status
- Test the entire management process through simulation using defined performance measures
- Identify and reconcile trade-offs

Spatial domains

One possible starting point for defining spatial management units is adoption of currently designated Large Marine Ecosystems (e.g., Sherman and Duda 1999). An important advantage of the LME option is that clearly defined spatial domains have already been identified in the 66 designated LMEs. As noted above, preliminary ecosystem models have already been developed in each of these areas. Further subdivision of LMEs may be desirable to account for finer-scale productivity patterns or other considerations (e.g., Fogarty et al. 2011). In general, we can envision a nested hierarchical structure of spatial management considerations within LMEs. For example, protected areas designed to meet multiple ecosystem objectives, including protection of vulnerable habitats, biodiversity hotspots, and (or) concentrations of threatened or endangered species or stocks, can be nested within LMEs (Fogarty 1999). In ocean basins, additional spatial units will have to be specified, perhaps using deep-water portions of defined biomes (e.g., Longhurst 1998) or FAO statistical areas, etc.

Objectives and reference points

The central objective is to maintain system-wide productivity within defined bounds and establish mechanisms to protect individual ecosystem components. Decision rules might then be framed to ensure that the sum of the catches of individual species will not exceed a system-wide limit and that no species will be driven below specified threshold levels for each. The system-level constraint is based on estimates of productivity levels. This approach borrows from the "two-tier" management strategy established by the International Commission of the Northwest Atlantic Fisheries (ICNAF) for the Northeast US continental shelf (ICNAF 1974). A similar system-level constraint has been in place for the Bering Sea–Aleutian Islands fishery (BSAI) since 1984 (Witherill et al. 2000; D. Witherell, NPFMC, personal communication). System-wide "target" reference points can then be established to accommodate precautionary buffers to account for uncertainty. The protection thresholds for individual species[7] can be established based on knowledge of life history characteristics, insights from earlier single-species assessments and analysis, or on a purely precautionary basis. LeQuesne and Jennings (2012) show how insights into vulnerability can be obtained even in data-limited situations (see also Costello et al. 2012).

Additional objectives will be specified for non-harvested components of the ecosystem and ones subject to incidental catch (nontarget and protected species) or collateral damage (e.g., habitat). By-catch limits can be specified and counted against the allocation for targeted assemblages. More qualitative measures to protect habitats through the use of protected areas can be enacted if information on the relationship between habitat and productivity is not available. Finally, social and economic objectives can be specified in the context of the conservation objectives. For example, given two or more management procedures with comparable conservation benefits, we would seek one entailing the greatest social and (or) economic benefits.

Data requirements and tactical models

The approach taken to establish the system-wide productivity levels will necessarily be tailored to the available scientific information and resources. The approaches described in the section Models for EBFM can be used to set system-level "limit" reference points to guide management actions. Minimum data and monitoring requirements will include information on the catch, primary production (from satellites or other sources), or estimates of the abundance or relative abundance of the species in the assemblage. More extensive data resources will of course be required if more complex models are selected. Local experts will be in the best position to ascertain the most effective approaches and models for setting the upper catch limit under prevailing environmental conditions. In areas with a broader range of available modeling options, a multi-model inference approach would be desirable. In some cases, indicators may be the best choice to guide the establishment of the cap.

The original system-level cap for the northeastern US was established based on the results of an aggregate-species production model (Brown et al. 1976). The system-level limit for the BSAI was originally established on a precautionary basis based on examination of proposed allocations developed using single-species assessments (Mueter and Megrey (2006) subsequently re-evaluated the system-wide limit using an aggregate production model approach). In both the northeastern US and Alaska, the overall cap was approximately 25%–30% lower than the sum of the individual species MSY levels. In the northeastern US, the allowable catch for individual species was set using a linear programming approach incorporating penalties for by-catch. In the BSAI, catch allocations for individual species are determined by negotiation among stakeholders, provided that the upper cap is not exceeded; if agreement cannot be reached, the council makes the determination (D. Witherell, NPFMC, personal communication).

Simulation testing

Multispecies and full ecosystem models can be used as operating models to test performance of the proposed management procedure. It will be desirable where possible to employ several operating models for this purpose (Sainsbury et al. 2000). Testing the performance of the assessment model(s) and identifying potential weaknesses is critical. For example, Gaichas et al. (2012) used a multispecies model to test the performance of simpler assessment models employing different aggregation strategies for defining functional groups. Attempting to take the maximum total yield from the entire assemblage resulted in the collapse of

[7]There will be cases where in highly diverse systems that information on individual species is simply unavailable. Indeed, one motivation for the use of aggregate species models is to address this situation.

approximately 40% of the species in each of two different systems (Georges Bank and Gulf of Alaska). However, reducing exploitation to a level resulting in 90% of the maximum total catch provided a very sharp reduction in the number of species being driven to collapse (to 10% or less of the species).[8] The remaining species still in trouble are predictably those with "slow" life histories and these will require additional forms of protection. Worm et al. (2009) show very similar results for Georges Bank using a length-structured multispecies operating model. These analyses reveal the dynamic tension between maintaining biodiversity and extraction of yield and point to the utility of adopting a precautionary harvest level (see also Brander 2010). Performance measures based on the distance between an indicator level and a reference point are vital in assessing the overall success of the management process.

A number of different incentive–disincentive structures can be put in place to adjust the exploitation patterns as part of the overall management procedure. For example, tax (Appolonio and Dykstra 2008), tariff (Kraak et al. 2012), and point (Anderson 2010) systems have been proposed to influence exploitation patterns. They can be used to "nudge" exploitation rates away from critical levels for vulnerable species.

Trade-offs

The trade-offs that emerge naturally when we adopt the EBFM perspective are typically not taken into account in conventional management. Unfortunately, trade-offs do not go away when ignored. They do, however, lead to suboptimal decisions and outcomes. We can readily define the trade-offs involved in many instances, but the information on societal preferences that managers need to attach weights to different courses of action is often lacking. There is of course a well-developed framework for coping with trade-offs arising from competing objectives, dealing with uncertainty, and explicitly incorporating values and preferences in management decisions (e.g., Keeney and Raiffa 1993). It is clear that there is considerable value in following a formal decision-theoretic process to frame the problem and its dimensions even if a satisficing solution is ultimately chosen. The decision-theoretic framework goes far beyond simply identifying that conflicts and trade-offs exist. It essentially entails (*a*) specifying a set of policy alternatives for a carefully bounded problem, (*b*) defining a set of attributes against which management actions will be evaluated, (*c*) assigning weights to the attributes that reflect both objectively defined characteristics and values and preferences, and (*d*) assigning each policy alternative a score against each attribute (Healey 1984). Adopting a decision table framework (Hilborn and Walters 1992) can be invaluable in understanding trade-offs.

Summary

In their seminal monograph, Beverton and Holt (1957, p. 24) called for "… the investigation not merely of the reactions of particular populations to fishing, but also of the interactions between them and the response of each marine community to man's activity". Our current single-species approaches maintain a convenient fiction: that we can keep individual species at biomass levels supporting single-species MSY (or related reference points) while ignoring interactions among species and environmental change. These approaches do provide clearly defined reference points and their implementation has helped significantly in controlling fishing pressure, which is critical to rebuilding depleted stocks. But they mask an inconvenient truth: that to the extent that MSY can be specified for an individual species, it is conditioned on the abundance of other species, management actions affecting these species, and changes in ecological and environmental conditions (including climate change). From this perspective there is no fixed single-species MSY — it rests on a multidimensional surface that is continually changing.

A commonly voiced concern is that the scientific, analytical, and regulatory frameworks for EBFM (and EBM) remain untried and therefore risky. It must be recognized that pathways toward EBFM are steadily evolving and will continue to develop as challenges are successively identified and solutions found. The current single-species management approach of course underwent a comparable development and evolution (Hilborn 2012). If we had waited until all issues and uncertainties had been resolved before implementing rigorous single-species management, we would now be facing much greater problems in the state of world fisheries. It is not necessary that we possess full knowledge of ecosystem structure and function before acting to incorporate ecosystem principles in fishery management if appropriate precautionary measures are adopted.

When contrasted with our ability to assess the status of individual species or stocks on a global basis, relatively simple multispecies and ecosystem models offer opportunities for broader coverage of fisheries systems. Detailed stock assessments are currently possible for only a small fraction of exploited fish populations (Costello et al. 2012), and most of these are concentrated in the developed world. Some of the simpler community-level or ecosystem models and approaches described above may offer avenues to addressing this problem. The proposed focus on maintaining diversity in these systems can address the recognized problems that accompany highly selective fishing patterns under conventional management. These practices often inadvertently result in imbalances in system structure and other problems (Fogarty and Murawski 1998; Zhou et al. 2010; Rochet et al. 2011; Garcia et al. 2012). Tactical tools for EBFM must be selected that avoid similar unintended consequences. Fogarty and Murawski (1998) noted that species-selective harvesting and discard practices that ignored community and ecosystem structure resulted in dramatic changes in fish community composition on Georges Bank and called for "… harvesting patterns encompassing a broader suite of species at much lower exploitation rates than at present". Garcia et al. (2012) identify the need for "balanced" harvesting strategies, echoing concepts developed by Swingle (1950) for freshwater systems.

I have primarily concentrated on possible solutions to concerns related to the natural science dimensions of EBFM. Many of the reservations concerning the feasibility of implementing EBFM have arisen in this sphere. However, we cannot lose sight of the fact that fisheries represent a ubiquitous form of social-ecological system involving a diverse set of physical, biological, economic, cultural, and governance considerations. They are best considered as complex adaptive systems (e.g., Allen and McGlade 1987; Liu et al. 2007; Gaichas 2008). Sudden shifts in state are a hallmark of such systems (Holling 2001; Mangel and Levin 2005; Mullon et al. 2005; Vert-pre et al. 2013) that require careful attention to the interplay of both social and ecosystem dynamics. Glaser et al. (2013) suggest that the layered complexity of fishery systems is evident in the higher incidence of nonlinear dynamics in metrics of fishery performance (e.g., catch or landings) relative to underlying ecosystem metrics as revealed by nonlinear time series analysis. These features are not captured in conventional assessment and management approaches that almost invariably consider fishery systems as involving a one-way interaction between humans and fishery resources and characterized by globally stable equilibrium points. In this case, choices of model structures can

[8]Collapse was defined in this study as reduction to below 10% of the maximum population level. In practice we would carefully consider the threshold level for collapse and likely choose a more precautionary level.

sharply constrain understanding. Adoption of EBFM, with its focus on humans as an integral part of fisheries ecosystems, provides a clear avenue for incorporating these perspectives and approaches into management. There can be little question that direct consideration of human motivations, needs, and values must be an integral part of the EBFM framework and a much broader adoption of strategies for co-management is essential if we are to avoid the past mistakes in management in which the human dimension was downplayed and the management problem was treated as "simple" ecological engineering (Charles 2001; Garcia and Charles 2008; Berkes 2012).

Rather than conceiving of fishery ecosystems as involving fixed-point equilibria (or even averages and fluctuations around fixed points) for individual species that are independent of other species and of the environment, we need to shift our focus to a perspective that seeks to provide sufficient resilience to allow the system as a whole to remain within stochastic bounds defined by past levels of variability. See Cury et al. (2005) for related discussions framed in the context of viability analysis, in which the preservation of viable (sustainable) ecosystem states remains the focal point for management decisions. We should replace the concept of single-species MSY, with its focus on time-invariant equilibrium processes, with a dynamic ecosystem yield concept that recognizes shifting environmental states and the probabilistic nature of production processes at different levels in the food web.

In framing the arguments presented above, I have of course done nothing more than to restate the insights and perspectives offered by a succession of commentators over the last several decades. It is long past time to act on their recommendation that we adopt a more holistic perspective in fisheries management. Because I have fallen into most of the traps I have described related to conventional modeling and management approaches, I am acutely aware of their allure, apparent justification, and the need to avoid them. While ecosystems are unquestionably complex, carefully chosen pathways toward EBFM can afford opportunities for simplification relative to management approaches now focusing on individual species or populations while side-stepping the limitations of single-species management.

Acknowledgements

I am very grateful for the thoughtful comments on this essay by Jeremy Collie, John Steele, and two anonymous referees. The views expressed here are my own and do not necessarily reflect the position of the National Marine Fisheries Service. I am deeply indebted to past and present members of the Ecosystem Assessment Program and other colleagues at the Northeast Fisheries Science Center who have generously shared ideas that have influenced my perspectives on this and related issues in ways direct and indirect: Richard Bell, Trish Clay, Geret DiPiper, Gavin Fay, Kevin Friedland, Sarah Gaichas, Scott Geiss, Jon Hare, Kim Hyde, Scott Large, Jason Link, Hui Liu, Sean Lucey, Marie-Caroline Martin, Matt McPherson, Bill Overholtz, Anne Richards, David Richardson, Vince Saba, and Laurel Smith. This paper is dedicated to the memory of John H. Steele. John graciously shared many ideas that helped immeasurably to shape this essay. Errors of interpretation, omission, or commission are of course mine alone.

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