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Viability theory for an ecosystem approach to fisheries

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Maintenance of overall ecosystem complexity is perceived as critical to the sustainability of ecosystem use. The development of an operational basis for an ecosystem approach to fisheries, however, faces many difficulties. On the research side, the challenge is in defining proper long-term, ecosystem-related objectives; determining meaningful reference values and indicators for desirable or undesirable states of the ecosystem; and developing appropriate data collection, analytical tools and models. The "viability" concept developed in economics by Jean-Pierre Aubin can be used to assist in the definition, selection of, and interaction among long-term objectives at an ecosystem level. It recognizes that ecosystems are complex assemblages of interacting and self-organizing natural and human components that cannot be predicted. Viability models define an ensemble of "viable states", in contrast to undesirable states defined as such by ecological, economic, and/or social constraints. These constraints can be derived from fisheries objectives, conservation principles, scientific results of modelling, or precautionary principles, and correspond to limit reference points to be avoided. Viability theory does not attempt to choose any "optimal solution" according to given criteria, but selects "viable evolutions". These evolutions are compatible with the constraints in the sense that they satisfy them at each time and can be delineated by the viability kernel. The southern Benguela marine ecosystem is presented as a first attempt for the application of this theory. In defining ecosystem-based objectives (and related issues such as target reference points), it seems more difficult to reach consensus among stakeholders on what is desirable than on what is undesirable (e.g. biological or economic collapse, species extinction, displacement of local rural communities). Expressed in the negative form or as limit reference points, ecosystem-based constraints can be considered simultaneously with current target reference points, such as maximum sustainable yield, using viability models. The viability approach can help to progressively integrate ecosystem considerations, such as conservation, into fisheries management.

Keywords: ecosystem approach, fisheries management, indicator, models, prediction, viability.

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Introduction

The quasi-general failure of conventional fisheries management is generally recognized (Garcia and de Leiva Moreno, 2003). Sources of failure are numerous and may not be exactly the same for all fisheries. In general, however, failure can be tracked to ineffective governance based on poorly communicated or simply inadequate science that fails to take multispecies effects, ecosystem effects, and/or effects of climate variability into account. In the developing world, fisheries science has often been underfunded and deeply skewed towards biology using scarce data of poor quality. Everywhere, scientists necessarily have oversimplified the nature and dynamics of resource and neglected the socio-economic dimensions of fisheries. As a result, they have provided elaborate management advice that has often been impractical, in many cases not followed by decision-makers, and in most cases poorly implemented. The overall consequence has been a series of ecological and economic collapses in numerous important fisheries in many parts of the world (Pauly *et al.*, 2002).

Fisheries are deeply imbedded within ecosystems, which are now rightly viewed as an integrative level for fisheries management, and its overall complexity is perceived as critical to its sustainability. The effects of fishing on marine ecosystems have been widely recognized (Hollingworth, 2000), as has been the need to move towards an ecosystem approach to fisheries (EAF; Sinclair and Valdimarsson, 2003). EAF, which deals with ecosystems instead of individual stocks, land-based pollution, environmental variability, in addition to globalization and fishing capacity reduction, and decentralized instead of top–down decision-making, is a more complex and likely more costly exercise than the conventional approach (Garcia and de Leiva Moreno, 2003). The question, however, is not whether EAF is required but whether the relevant authorities will be able to implement EAF successfully before society decides that fisheries are too costly and unsustainable to be worth being maintained (Cury and Cayré, 2001). It is clear that we need to move towards an EAF, but the first question is why, and how, a much more complex approach might have greater success than single-species management has had.

We present the challenges faced by scientists in addressing the growing societal requirement to move to EAF with all the added difficulties associated with developing and promoting new operational guidelines. We discuss the recent ideas that have been developed to adapt conventional concepts or to shift more radically to new paradigms. As a contribution to this effervescent scientific debate, we introduce the viability theory, first developed in mathematics and later applied to economy (Aubin, 1991, 1996). We propose viability as a valid principle for EAF, then briefly describe how viability models can be used as tools to explore and simplify complex ecosystem dynamics defining multidimensional spaces containing viable management strategies. Within this context, we discuss the use of Limit Reference Points (LRP) to guide and assess performance of management strategies and policies, as opposed to Target Reference Points (TRP). Finally, we examine how viability theory could help to turn modern management-orientated fishery science into a more "falsifiable" discipline.

Towards a challenging EAF

Conception of an operational EAF faces many issues, ranging from the high cost of the science required to the practical difficulties of changing the governance system and processes. From a scientific perspective, difficulties are related to: (i) defining proper long-term, ecosystem-related, objectives; (ii) determining meaningful indicators and reference values for desirable or undesirable ecosystem states; and (iii) developing appropriate data collection, analytical tools, and models.

The lack of clarity in, and contradictory nature of, fishery management objectives, to the extent these have been made explicit, has been a serious problem in conventional fisheries management (Sainsbury *et al.*, 2000). It is even more difficult to define clear operational objectives for use in an ecosystem context than it is for more tractable ecological entities such as stocks, whether single or multispecies. Pitcher and Pauly (1998) suggest that "rebuilding ecosystems, not sustainability" should be the proper goal of fishery management. For other authors, promoting ecosystem integrity is of high priority. All of these otherwise pleasant-sounding objectives are poorly defined or misnomers, according to Link (2002a). Because ecosystems may exhibit multiple states and regime shifts, rebuilding targets are difficult to define, and therefore "ecosystem health" or "ecosystem integrity" cannot be measured and monitored. Long-term, practical objectives for EAF are strongly debated (Constanza *et al.*, 1997), and scientific consensus has not yet been achieved.

Models available to reproduce and simulate ecosystem dynamics are based on far-reaching ecological assumptions and may not take account of parameter uncertainty (Whipple *et al.*, 2000). Also, they are often constrained on spatial and/or temporal scales and are, therefore, not sufficiently dynamic. Problems related to scale-dependency may severely hamper model comparisons among ecosystems or even between different states of the same ecosystem over time, so limiting their usefulness in decision-making. Inconsistencies in data availability and quality may cause bias. For example, more data, often of higher quality, are usually collected for commercially important species, compared with those species that are not perceived as economically valuable, but may well play crucial roles in ecosystem functioning. Scepticism has been expressed repeatedly about the realism of the models currently used to analyse properties of foodwebs and ecosystem dynamics (Jennings *et al.*, 2001). It also remains doubtful that available ecosystem data and models are always adequate. Despite valuable attempts to make models of trophic interactions more dynamic (Walters *et al.*, 1997), the difficulty to tune these properly with existing data remains a concern. The potential of available ecosystem models to make realistic predictions that could be used for management purposes remains untested (Mace, 2001).

A shift in paradigm away from single-species management is required (Christensen *et al.*, 1996), but this certainly necessitates integrating previously accumulated scientific results and management expertise (FAO, 2003). Link (2002a) views this transition as an iterative process that facilitates a dialogue to clearly define ecosystem-related fishery goals, as well as negotiation protocols to resolve conflicts between competing uses. Notwithstanding, a radical change towards EAF requires an innovative strategy to define reference points based on suitable ecosystem metrics and to bridge the gap between scientific results, social needs, economic goals, and effective fisheries management. The lack of well-defined long-term objectives, of appropriate tools, and of consensus on which indicators to use, can, however, jeopardize the present momentum towards improving fisheries management at an ecosystem level and could lead to an over-simplistic, and likely unsuccessful, implementation of EAF. Therefore, new avenues need to be explored in which the absence of crisp objectives and accurate ecosystem models is less of a problem.

The viability principle

Viability as an objective

Nature may not be predictable, but it is not totally unpredictable either. Ecosystems are structured and, as a consequence, strong patterns of interaction emerge (Cury *et al.*, 2000, 2003). These patterns should be taken as the entry point to EAF, even if the lack of a general theory on the functioning of marine ecosystems results in poor predictive power. The transient nature of strong patterns should be recognized, together with the fact that ecosystems can potentially exhibit states that may be undesirable from the point of view of fisheries management (such as shifts in dominance between demersal fish and pelagic fish resources).

There appears to be more experience with, information on, and consensus about undesirable states of ecosystems than there is about theoretical ‘‘optimum’’ states. Historical examples of overexploitation and degradation of ecosystems, and the societal reactions to these, have led to a widespread acceptance of LRPs as borders of undesirable states (e.g. minimum spawning biomass necessary to maintain recruitment) compared with TRPs as indicating specific desirable states (e.g. optimum biomass of small pelagic fish to be left as forage for predators; Caddy, 2002). Therefore, although we may not know enough about ecosystem functioning to develop models that aid us in optimizing the contribution of ecosystems to society, we may know enough to model (and provide advice on how to maintain) the conditions under which ecosystems are likely to persist as sources of goods and services.

Link (2002a) proposed to aim for ecosystem persistence (or ‘‘ecosystem state sustainability’’) through maintenance of specified processes. Recognizing the complexity of exploited living systems, Bossel (2001) argues that management performance indicators must reflect management effectiveness in ensuring the viability of essential component systems (or subsystems), as well as of the whole ecosystem. This is the focal point of what could be named a viability approach, which implies certain flexibility in human actions, as well as uncertainty in our understanding, evaluation, and prediction of the system dynamics. A viability approach recognizes the indeterminacy of natural systems and the lack of well-defined management objectives. In a system where uncertainty is overwhelming, and taking a single-species approach to fisheries management, the best strategy might not be to optimize the goods and services that can be extracted from the different species available, but rather to define the range of catches of the different species that can be extracted without compromising overall ecosystem dynamics.

Integrating a set of constraints, which seems critical when defining the ‘‘health’’ of an ecosystem, appears to be easier to achieve than assembling a set of targets. In fact, fisheries scientists have progressively attempted to integrate sets of constraints into both single-species and ecosystem-based models. Figure 1 illustrates the different paradigm shifts, models, and indicators that have been or are used in fisheries management. After considering that marine resources were almost inexhaustible and that human activities had no major effects (Figure 1, A1), it was rapidly recognized that fisheries had a measurable impact on stock abundance, and it was believed that the catch of a given species could be maximized by adjusting fishing effort (maximum sustainable yield, MSY; Figure 1, A2). This view was challenged because of many assumptions, uncertainties, and poor evidence obtained for many stocks; the MSY concept *sensu stricto* has been criticized since the mid-1970s (Larkin, 1977). The concept, however, is enshrined in the Law of the Sea and has survived in international binding agreements. In recent years, a single-species precautionary approach has been adopted by ICES (1998), which uses sets of constraints to define ‘‘outside safe biological limits’’ (Figure 1, A3). The precautionary parameter space for ‘‘safe’’ exploitation in terms of spawning-stock biomass (B) and associated fishing mortality (F) is defined by estimates of LRPs (B_{lim}, F_{lim}), as well as by precautionary reference points (PRPs: B_{pa}, F_{pa}). While the LRPs should demarcate the true domain of sustainability, the PRPs take into account the uncertainty in current estimates and predictions. Effectively, recent TAC advice provided within the ICES community is therefore based on the avoidance of undesirable states (ICES, 2003).

Ecosystem models such as Ecopath−Ecosim (Walters *et al.*, 1997) and associated suites of indicators were developed, specifying the trophic interactions among different components of the ecosystem (Figure 1, B1). However, the lack of understanding and the unreliability of predicting ecosystem responses to fishing and other forcing factors (such as pollution, natural variability, or climate change), as well as the costs of alternative assessment strategies, have resulted in the continued use of conventional single-species management by default (Hoffman and Powell, 1998). The lack of ecosystem-based reference points led to proposals for assigning a new role to MSY. As reflected in the 1995 UN Fish Stock Agreement, MSY

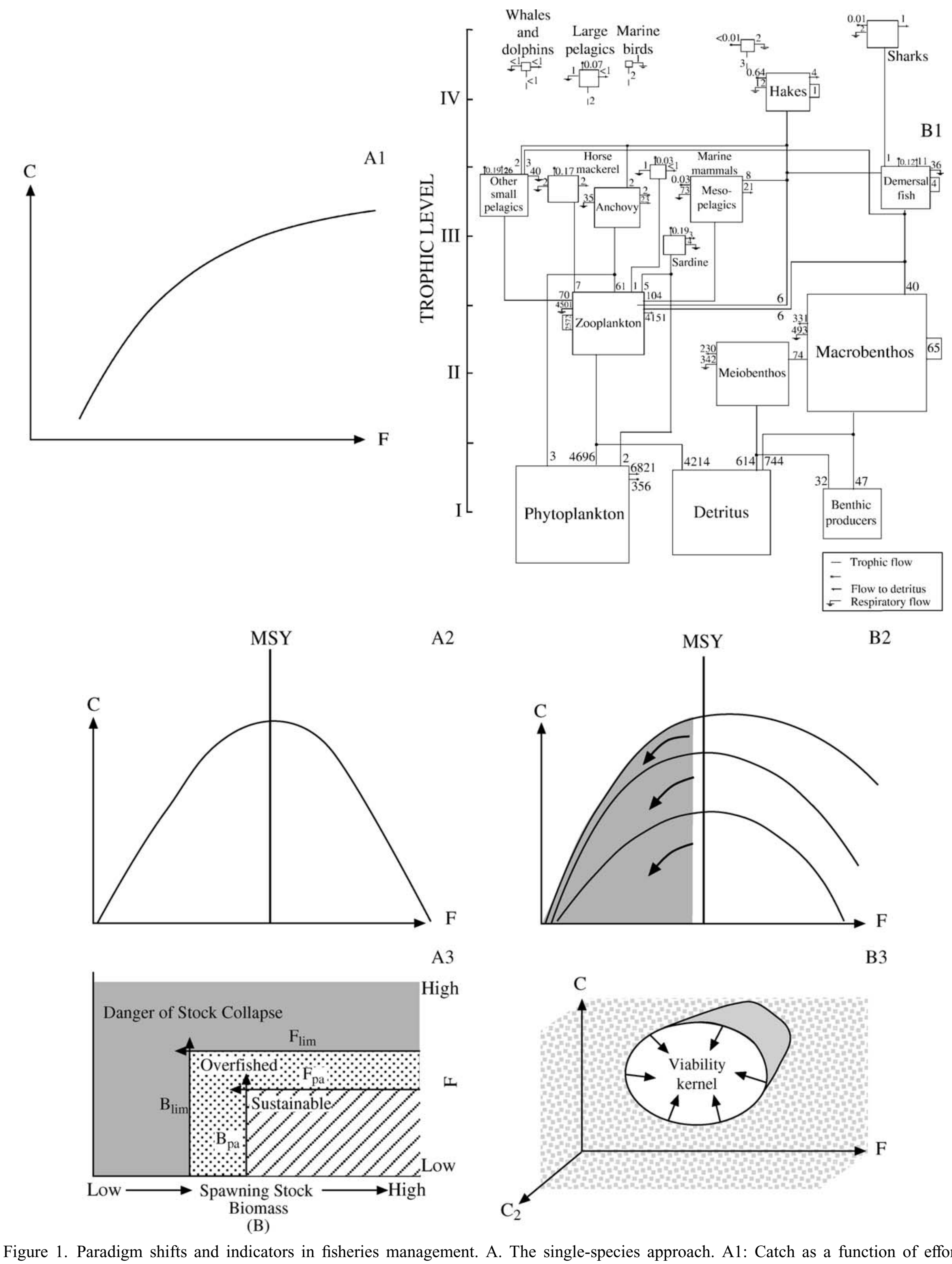

Figure 1. Paradigm shifts and indicators in fisheries management. A. The single-species approach. A1: Catch as a function of effort without constraints (Huxley, cited in Roberts and Hawkins, 1999); A2: The MSY concept (Schaefer, 1954). A3: Precautionary approach as developed by ICES (www.cefas.co.uk/fsmi/pa_management.pdf). B. Ecosystem approach to fisheries. B1: Ecopath/Ecosim modelling (Pauly *et al.*, 2000); B2: MSY as a limit reference point defining a viability domain (Mace, 2001); B3: Viability kernel that contains all viable trajectories of a complex system constituted of components in interaction (e.g. F, fishing effort; C, catch of predatory fish; C2, catch of prey fish; Mullon *et al.*, 2004).

should be considered as a biological limit that should not be surpassed, rather than as a development target (Figure 1, B2). The belief is that this use of MSY, if implemented, would imply such a significant effort reduction that ecosystem effects of fishing would be substantially reduced, including impacts on biodiversity and genetic diversity (Mace, 2001).

The viability approach is aimed at contributing to this development by defining boundaries in a more rigorous way. It recognizes that numerous components in interaction have to be considered simultaneously, and that undesirable states exist and should be avoided. By considering a set of constraints, it is possible to define all viable trajectories of the dynamic system (Figure 1, B3). Therefore, the objective is not to optimize one or several variables, but to stay within the viability kernel, ensuring acceptable trajectories within a dynamic system. The viability approach forces us to define, *a priori*, long-term objectives in a sense that clear definition of the constraints to be avoided is required. In essence, the viability approach parallels the development in single-species management advice taken by ICES (1998), by defining limits for sustainable development but extending this to the ecosystem level.

The constraints to define the long-term, perennial objectives in the EAF may be ecological, economic, and/or social. Using the viability approach requires that: (i) objectives adopted be conditioned by the viability of the whole ecosystem, as defined by quantitative constraints on its components (i.e. none of them reaches predefined, undesirable states at any time); (ii) interactions governing the ecosystem dynamics be known, together with associated uncertainties; (iii) the viability kernel accounting for uncertainty and delineating all viable trajectories can be calculated; (iv) management options be expressed in such a way that they can maintain the system within the viability kernel.

Viability theory as a modelling tool

The viability principle has received a formal mathematical treatment known as the "viability theory" (*sensu* Aubin, 1991, 1996). A viability model is an attempt to describe possible evolutions of a dynamic system under uncertainty compatible with constraints on its state variables, delineating conditions considered to be undesirable, or fulfilling dynamic constraints implemented in the model. Viability models do not lead to optimizing a time-related criterion as in optimum control theory, but instead define all viable evolutions of a dynamic system under uncertainty, satisfying at each instant specified constraints, which might only be known approximately. These evolutions, delineated by the viability kernel, are compatible with the constraints in the sense that they satisfy them at all times.

The theory assumes that the evolution of many variables describing systems, organizations, and networks arising in biology, human, and social sciences do not evolve in a deterministic way, and possibly not even in a stochastic way, as usually understood. Instead, it assumes a Darwinian flavour, where time-related optimality selection mechanisms are replaced by several forms of "viability requirements", a term encompassing polysemous concepts such as stability, confinement, and homeostasis, expressing the idea that certain variables must obey certain constraints. Time-related optimization is replaced by myopic selection mechanisms that involve current knowledge, and sometimes knowledge of the past, instead of anticipation or knowledge of the future. Unpredictable rare events (natural perturbations or disturbances) that obey no statistical law must be avoided at all costs (precautionary principle or robust control). These systems can be regulated by use of controls that have to be chosen to guarantee their viability or the achievability of targets and objectives (Aubin, 1996).

The theory was motivated by dynamic economics out of equilibrium and Darwinian evolution and has been developed mathematically since the beginning of the 1980s, with few applications to ecological systems (Bonneuil and Mullers, 1997; Lefur *et al.*, 1999; Béné and Doyen, 2000; Béné *et al.*, 2001). Mullon *et al.* (2004) recently proposed a viability approach to trophic interactions in the exploited southern Benguela marine ecosystem as an alternative to classical ecosystem models. A brief summary is presented here (for details of the model and the iterative algorithm, see Mullon *et al.*, 2004). In this simple model (Figure 2), the ecosystem consists of five components interacting dynamically under a set of constraints (*in casu*, threshold biomasses for all components; Table 1). Fishing is formulated as potential yield (e.g. MSY) for both the pelagic and the demersal fish components. Interactions between the components are expressed on the basis of trophic coefficients derived from a mass-balanced Ecopath model for the ecosystem (Shannon *et al.*, 2003). The associated variability of the coefficients is derived from various estimates and through the use of the Ecoranger routine of the Ecopath model. This approach takes into account several specificities of real ecosystems, such as predator or donor controls of one component by the other.

Thanks to the linear structure of the viability model, an iterative algorithm can provide an approximation of the viability kernel. The kernel, containing all possible combinations of biomasses of the five interacting components potentially coexisting in the system, necessitates a 5D representation. As there is no straightforward way to represent such a multidimensional kernel, an example is presented here of several 3D slides of the zooplankton/demersal fish/pelagic fish biomass kernel at different levels of detritus and phytoplankton biomass (Figure 3). In this particular example, the yields of pelagic and demersal fish have been assumed to be 3 and 2 $\mathrm{t\,km^{-2}}$, respectively.

The main features are that the system is not viable below a threshold value for detritus of about 1200 $\mathrm{t\,km^{-2}}$, and always viable above a threshold value of about 1600 $\mathrm{t\,km^{-2}}$, and that in the critical strip between these

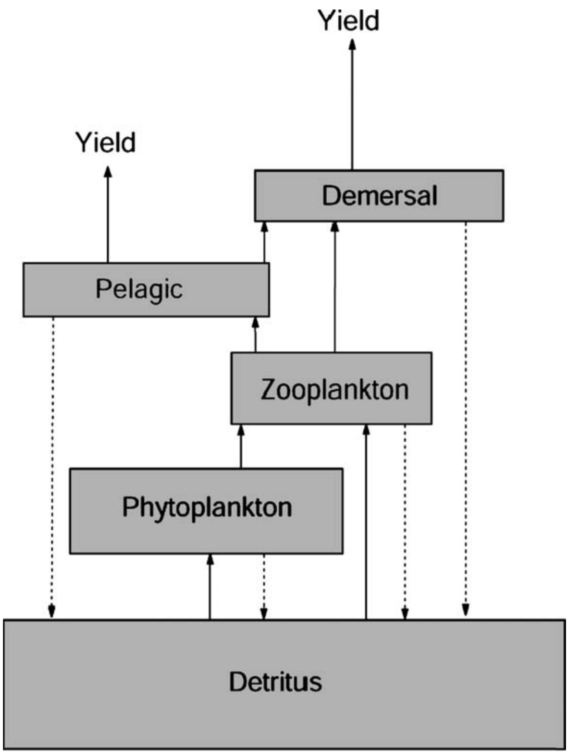

Figure 2. Trophic interactions between five components of the southern Benguela ecosystem (adapted from Mullon *et al.*, 2004).

two thresholds and for low values of phytoplankton, a combination of zooplankton, pelagic fish, and demersal fish is sufficient and necessary to ensure the existence of a viable trajectory. At this stage, and knowing the uncertainty of several parameter values, this example must be considered as experimental. Nevertheless, the fixed yields of both pelagic and demersal fish appear realistic for high values of detritus and phytoplankton biomass.

The constraints and strength of the interactions define the viability of the ecosystem configurations. This has several implications. Constraints are formulated in a negative form as things we do not want to see happening (i.e. using LRPs). As constraints represent limits beyond which the necessary conditions for either human or environmental well-being are not satisfied, it is postulated that considering them is vital in achieving the purpose of EAF. Because the model can be run with data currently collected for assessment and management purposes, and constraints can be calculated at different levels of knowledge − from well-documented to poorly known systems, and at different scales of space and time − comparisons may be made among ecosystems, ecosystem states, or time periods. Non-viable trajectories are more heavily weighted than viable trajectories. There are several reasons for this. First, adding more constraints might drive the system into non-viable states: the viability kernel gets smaller as the number of constraints increases. Second, more importance is assigned to the avoidance of undesirable states, which is more precautionary and easier to attain than attempting to reach a particular target. Finally, constraints proposed must comply with management and conservation objectives and strategies.

Table 1. Ecological constraints in terms of biomass (B_{min} and B_{max} defined arbitrarily in t km^{-2}) on the state of each component for the viability model of the southern Benguela ecosystem (from Mullon *et al.*, 2004).

Compartment	B_{min}	B_{max}
Detritus	100	2 000
Phytoplankton	30	400
Zooplankton	20	200
Pelagic fish	5	60
Demersal fish	5	30

Presentation of the results in the form of a viability kernel facilitates recognition that other possible constraints may exist besides those that have been considered in the model. This means that results can be discussed and put in perspective within a multidisciplinary framework of representation (sustainable development reference system; Garcia and Staples, 2000). Also, it can be used as an ecosystem-based indicator for fisheries management that can be understood and accepted by policy-makers and the public at large. Because of the complexity of the calculations required to define the viability kernel, only a limited number of constraints and interactions can be taken into account (currently nine), but new algorithms are under investigation that will allow consideration of more interactions simultaneously and the development of ways to represent them.

Discussion

Assembling our disparate knowledge into an operational ecosystem-based framework is an immense task requiring time, resources, and new approaches. Given the available knowledge, reaching consensus on ecosystem-based objectives and target reference points for fisheries management among numerous stakeholders will not be easy. Social, economic, and ecological objectives are likely to be contradictory, and decisions will be required at the local scale on how best to reach a workable balance. In addition, if agreement is reached among diverging views and interests, the compromise objectives are often expressed in very general (and often rather useless) terms, and/or are unachievable in practice. The move to EAF requires a shift, or at least a serious adjustment, from the conventional paradigm of decision-support science, as prediction of complex ecosystem dynamics is out of reach. Several attempts are made to achieve viable states for fisheries in marine ecosystems, as illustrated by the evolution of ideas, indicators, and models. This shift might be to a more

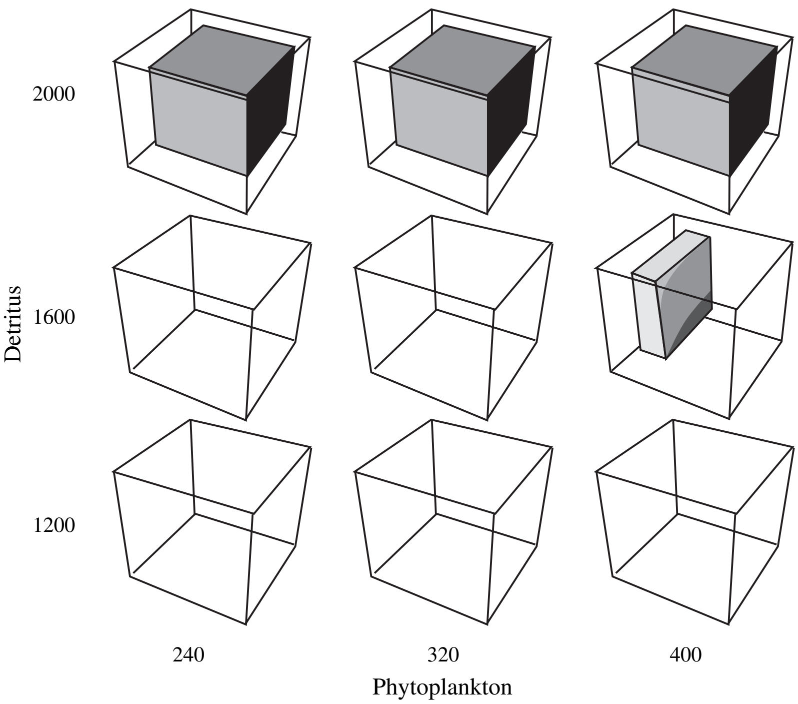

Figure 3. Viability kernel approximation to the southern Benguela ecosystem (grey; horizontal axis, zooplankton biomass; vertical axis, demersal fish biomass; depth, pelagic fish biomass; adapted from Mullon *et al.*, 2004) for different levels of detritus (ranging from 1200 to 2000 t km^{-2}), and phytoplankton (ranging from 240 to 400 t km^{-2}; for constraints see Table 1). Yields of pelagic and demersal fish are 3 and 2 t km^{-2}, respectively.

adaptive management procedure, which allows a system to be monitored continuously and objectives constantly updated as we learn from what should be avoided. Consequently, EAF should be based on more easily agreeable limit reference points, i.e. used as barriers to maintain or reconstruct the ecosystem and the fisheries it sustains, and to keep it away from undesirable states. Recognizing and understanding limits is important so as not to incur the risks and loss of integrity when limits are exceeded (Fowler and Hobbs, 2002).

We believe that viability theory can contribute to reconciling different management approaches as well as different management objectives. There is fear that including ecological considerations may increase complexity and thereby add further uncertainty to the management process (Link, 2002b); instead we think that the viability approach can rejuvenate old reference points. Classical approaches to fisheries management could benefit because viability theory can help to reconcile formerly agreed objectives to ecosystem-based considerations. It can also help to integrate conservation, as well as social or economic issues that are not usually incorporated into conventional fisheries management. Applications still need to be generalized and standardized, and computing methods need to be further refined; these developments may still be a long way off. Viability theory as developed by Aubin (1991, 1996) and his collaborators appears, however, to be a powerful tool for incorporating sustainability constraints into a dynamic model. If it receives adequate attention, it could generate a new field of research for providing management advice, as well as a framework for defining "ecosystem health". As such, viability theory might represent a key contribution to EAF.

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